United States
Department of
Agriculture

Forest Service

**Rocky Mountain
Forest and Range
Experiment Station**

Fort Collins,
Colorado 80526

**General Technical
Report RM-239**

Management Impacts on Water Quality of Forests and Rangelands

Dan Binkley
Thomas C. Brown

Overview

This report compiles information about the effects of management practices on water quality in forests and rangelands. Chapter 1 summarizes water quality concerns on all types of lands. It discusses categories of water quality degradation, land area sources of degradation, processes and activities that cause such degradation, and recent trends in water quality in the United States. Chapter 1 places water quality problems on forests and rangelands in the context of the broader concern. In most cases, forest and rangeland management activities are relatively minor contributors of water quality degradation.

Chapters 2 through 9 focus on physical, chemical, and biological aspects of the forest and rangeland water quality. Chapter 2 reviews basic forest and rangeland hydrology and water quality processes, and it ends with a brief description of where we obtained the water quality data for this report. Chapters 3 through 8 review the state of knowledge about the effects of land management actions on water quality of forests and rangelands in 6 regions of North America. Each of these regional chapters focuses on results at experimental watersheds within the region. Each chapter ends with a short summary; a table summarizing the findings at the region's experimental watersheds is in the Appendix. Chapter 9 provides a synthesis of the 6 regional chapters. This synthesis concludes that suspended sediment, especially in areas of sensitive soils and slopes, is the major water quality concern. Best management practices (defined in Chapter 10) generally minimize suspended sediment concentrations.

The scope of Chapters 3 through 9 is limited in two ways. First, the focus is on the effects of land management practices, such as harvesting and grazing, and not on the generally less important effects of acid precipitation on forest and rangeland water quality. Second, our emphasis is water quality, not erosion, so we do not review the many studies that have measured only on-land soil movement.

Chapter 10 describes the federal laws and state programs that are intended to control or monitor forest and rangeland management practices affecting water quality. The state programs are summarized in a table based on recent interviews with state personnel. Examples of state efforts are given. Then we discuss the rationale for basing nonpoint source pollution control on specification and use of best management practices. Carefully designed best management practices are encouraged, but cost effectiveness is also emphasized.

The final chapter reviews available information about the economic efficiency of nonpoint source pollution programs on forestlands. While the costs of implementing best management practices are fairly well understood in some locations, the benefits of their application are poorly documented. Better studies are needed to determine the efficiency of best management practices.

USDA Forest Service
Gen. Tech. Rep. RM-239

November 1993

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Dan Binkley
Colorado State University

Thomas C. Brown
Rocky Mountain Forest and Range Experiment Station[1]

Acknowledgments

We thank Jeff Jones for assistance in many ways; Xiaoming Zou and Jordi Cortina for help with the tables; and Joyce Hart for clerical assistance. We are also grateful to the following persons for their helpful reviews: Robert Beschta, Al Covich, James Gregory, Adrian Haught, Al Lucier, John Stednick, and Robert Stottlemyer.

[1] *Headquarters is in Fort Collins, in cooperation with Colorado State University.*

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Dan Binkley and Thomas C. Brown

Chapter 1

Introduction

Since passage of the Clean Water Act in 1972, considerable progress has been made in reducing municipal and industrial releases of water pollutants. This progress has allowed more attention to focus on contaminants that reach the nation's rivers and streams in runoff from many types of land, including forests and rangeland.

Concerns about the effects of forest management on water quality are not new (Kittredge 1948). In the United States in the late 1800's and early 1900's, conservation advocates were concerned about soil erosion and its impacts on water quality. For example, Gifford Pinchot (1910), an early Chief Forester of the USDA Forest Service, wrote: "The waste of soil is among the most dangerous of all wastes in progress in the United States. ... In the upland regions of the states south of Pennsylvania three thousand square miles of soil [have] been destroyed as the result of forest denudation. ... The soil so lost ... becomes itself a source of damage and expense, and must be removed from the channels of our navigable streams at an enormous annual cost." Furthermore, "The destruction of forage plants by overgrazing ... is accompanied by loss of surface soil through erosion; by forest destruction; by corresponding deterioration in the water supply ..."

Not all observers were convinced of the potential detrimental effect of vegetation removal on water quality. Early in this century, as forests were being added to the nation's system of forest preserves in order to limit timber harvest and secure "favorable conditions of water flow," the Chief of the U.S. Geological Survey stated, "What man does with forests will have little effect on erosion" (Kittredge 1948:13).

Such controversy contributed to establishing experimental forested watersheds where the impacts of management were, and are, studied. In response to concerns about flooding and downstream water supplies, studies at the experimental areas initially emphasized the effects of harvest on the quantity and timing of flows. Only in the past 25 yr or so, water quality and nutrient cycling have also received significant attention at the experimental forests. Water quality has also been studied at many other forest and rangeland study areas. Through this research, we have substantially increased our understanding of the potential effects of forest and range management on water quality. This report documents the findings at many of the experimental watershed areas in North America where the effects of land management on water quality have been studied. It also describes the laws and procedures in the United States that regulate forest and rangeland management in order to protect water quality.

Overview of Water Quality Concerns

Like "wildlife," "water quality" refers to a long list of individual components. Water quality is a function of a series of physical (e.g., suspended sediment), chemical (e.g., nitrate), and biological (e.g., giardia) constituents or indicators. Many of these constituents have common levels that occur under typical natural conditions, but the levels can be greatly affected by both natural events and human actions. For example, although some suspended sediment is natural in any stream, the level of suspended sediment can (depending on the weather) increase greatly after natural events such as wildfires or human actions such as timber harvest. Furthermore, human actions can lower concentrations below the natural level, as prolonged fire protection has done to selected constituents in many forest areas.

Water "pollution" typically refers to water quality degradation caused by human actions or decisions. Water quality degradation can of course be caused by natural events such as heavy rains or volcanic eruptions as well. Pollution is more likely to occur following a management action if the natural concentration level is already untypically high. Pollution occurs when the level of a parameter reaches the point where water users of importance (such as humans or fish) may be adversely affected. The incidence of pollution thus depends on which water users or uses are considered important. In some water quality assessments, the featured water users are not specifically stated (leading to some ambiguity), while in others they are "designated." For some uses, the levels at which some constituents or indicators become harmful have been agreed upon (water drinking standards are an example). For other uses or constituents, consensus is lacking on when pollution begins.

To bring some order to the complex arena of water quality and help place forest and rangeland water pollution problems in context, we first categorize the types and sources of water pollution.

Categories of water pollution

Seven categories of water pollution are listed in table 1, along with their respective water quality constituents or measures. Most of these constituents occur naturally in rivers and streams, and management activities can increase or decrease their levels. Also note that the levels at which problems emerge vary widely among the constituents.

Pathogenic organisms include certain bacteria (indicated by measures such as fecal coliform and fecal streptococcal bacteria), *Giardia* (a protozoan), harmful viruses, and certain fungi. They are waterborne disease-causing agents. Once a principal concern of water quality engineers and public health officials, this category of water pollution has been largely controlled in most areas of developed countries by effective water treatment and distribution procedures.

Organic material, from waste products and decaying plants, requires oxygen because it is decomposed by bacteria. This decay process creates "biochemical oxygen demand," lowering the dissolved oxygen available for fish and aquatic invertebrates to potentially lethal

Table 1.—Major categories of water pollution and related water quality constituents.

Category	**Principal constituents or measures**
Pathogenic organisms	Fecal coliform bacteria Fecal streptococcal bacteria
Organic material	Biochemical oxygen demand (BOD) Dissolved oxygen

Nutrients	Nitrogen
	  Nitrate
	  Nitrite
	  Ammonia
	Phosphorus
	  Dissolved ions and inorganic molecules
	  Particulates suspended with sediment or organic material

Suspended sediments, silts	Total suspended solids Turbidity

Dissolved solids	Specific ions
	  Sodium
	  Potassium
	  Calcium
	  Magnesium
	  Carbonate
	  Bicarbonate
	  Chloride
	  Sulfate
	  Iron
	  Manganese
	  Dissolved organics
	General measures
	  Total dissolved solids (TDS)
	  Conductivity

Toxics	Metals
	  Cadmium
	  Lead
	  Arsenic
	  Zinc
	  Copper
	  Chromium
	  Nickel
	  Mercury
	  Pesticides (insecticides, herbicides, fungicides)
	  Other (mainly organics such as polychlorinated biphenyls (PCBs))

Miscellaneous	Temperature pH

levels. It can also potentially cause water color changes and odor problems.

Nutrients primarily include forms of phosphorus and nitrogen. Sufficiently high nutrient levels cause excessive growth of aquatic plants and animals, leading to murky water, floating algae, and dense mats of aquatic plants. This condition is known as eutrophication and it degrades water use for recreation, fish, and wildlife. As the organic matter decays, it may in turn reduce dissolved oxygen. In addition, elemental phosphorus and nitrogen (such as nitrate, nitrite, and ammonia) can be directly toxic to fish and humans (nitrate and nitrite are especially toxic to infants).

Suspended sediments consist mostly of fine soil particles that are carried along in streamflow. Suspended sediments increase turbidity and transport plant nutrients, heavy metals, pesticides, pathogens, and other potential pollutants attached to the soil particles. Such particles can settle in streams and reservoirs, reducing water storage capacity, impairing fish habitat (especially spawning ability), and obstructing fish navigation as particles accumulate in water courses.

Dissolved solids include a series of ions (commonly called salts) as well as dissolved organic compounds. Total dissolved solids and specific conductivity are two measures of the overall concentration of these ions. Dissolved solids cause corrosion of pipes and water-using appliances, reduce yields of some irrigated crops, cause increased use of soaps and detergents, and can harm fish and other aquatic organisms at sufficiently high levels.

Toxics are chemicals that have adverse effects at extremely low concentrations. They include toxic heavy metals (e.g., mercury, lead, cadmium, arsenic) plus a host of synthetic, generally organic, pesticides and industrial materials (e.g., PCBs). With over 60,000 commercial chemical substances currently in use in the United States, the list of potential toxics is substantial. The impact of most in aquatic environments is unknown.

Finally, a *miscellaneous* category includes temperature and acidity. Temperature increases can alter habitat for fish and aquatic invertebrates. Increasing acidity (decreasing pH) mobilizes many elements, such as aluminum, which interfere with physiological processes of fish and other aquatic organisms.

Two recent national surveys indicate the relative prevalence of these categories of water pollution. The Environmental Protection Agency (EPA) biennially publishes an assessment of water quality in the United States. These assessments summarize state reports that are based on monitoring, surveys of scientists, predictive water quality modeling, and citizen input. The 1988 assessment (EPA 1990) reports data from about 35 states that have assessed about 25% of the nation's miles of rivers and acres of lakes and reservoirs. About 30% of the assessed river miles and 26% of the lake area were estimated to be affected by one or more categories of pollution to the extent that the "designated uses" (e.g., contact recreation, drinking water supply, high-quality cold water fishery) were not fully "supported." Excessively high nutrients was the most widespread problem, affecting 9% of the river miles and 14% of the lake area (table 2). Pathogens affected 6% of the river miles, while 7% of the lake area was affected by organic enrichment. Other categories affected less than 5% of the rivers or lake area. It should be noted that the EPA summary provides only a rough indication of the incidence of water pollution because of the variety of methods used by the different states and because the sample of assessed rivers and lakes was not systematically designed (some states did not report, and some of those that did focused on problem areas). Nevertheless, such surveys may provide useful indications of at least the relative importance of the different categories of water pollution.

The 1982 National Fisheries Survey (Judy et al. 1984) relied on state fisheries biologists' evaluations of a statistically based sample of 1,303 river reaches in the United States (roughly 10%). The biologists reported that "the survival, productivity, or use of the fish community [was] adversely affected" in 56% of the sampled stream miles (and 45% of the perennial stream miles). Suspended sediments and high temperatures were the most prevalent problems (table 2). The higher percentages for this survey, compared with the EPA summary, may be due to the exclusive focus on fish habitat in the fisheries survey rather than on selected "designated uses."

Sources of water pollution

Sources of water pollution are broadly categorized as point source or nonpoint source (table 3). The most important point sources are sewage treatment plants and industrial facilities. Effluents that cause point source pollution leave the source in a contained structure like a pipe or small canal. The water quality of these effluents can be monitored, and unacceptable quality degradation can be directly linked to the source. Nonpoint source pollution occurs as more diffuse runoff from land areas. The dispersed transport mechanisms of runoff make it difficult to monitor the degradation of water quality except at points downstream from the cause, where it is difficult to identify the activity or the specific land area from which the degradation originated.

Nonpoint source pollution can originate almost anywhere (table 3). Urban areas are sources of all the major categories of water pollution because runoff following precipitation events carries household products, pet wastes, yard applications, industrial chemi-

Table 2.—Incidence of water pollution in the United States according to two national surveys (% of assessed miles or acres).[a]

Category	Water quality inventory[b]		Fisheries survey[c]
	River and stream miles	Lake and reservoir acres	River and stream miles
Pathogenic organisms	6.3	2.5	—
Organic material	5.0	7.2	9.5
Nutrients	9.0	13.9	12.5
Suspended sediment	2.1	2.1	34.4
Dissolved solids	2.1	4.1	1.8
Toxics			9.8
Metals	3.7	2.1	—
Pesticides	3.5	1.5	—
Other	—	2.3	—
Miscellaneous			
Temperature	1.3	—	26.2
pH	1.7	1.5	3.0

[a]*The same miles of river or acre of lake may be affected by more than one category of pollution.*
[b]*Source: EPA (1990).*
[c]Source: Judy et al. (1984).

Table 3.—Land area sources and their principal categories of water pollution.

Land sources	Principal categories
POINT SOURCE POLLUTION	
Sewage treatment plants[a]	Pathogens, organic material, nutrients, toxics
Industrial facilities	Organic material, toxics
Thermal energy plants	Water temperature increase
NONPOINT SOURCE POLLUTION	
Urban areas[b]	Pathogens, nutrients, suspended sediment, dissolved solids, toxics
Farms[c]	Pathogens, nutrients, suspended sediment, dissolved solids, toxics
Mines[d]	Suspended sediment, dissolved solids, toxics, pH
Landfills	Toxics, etc.
Animal feedlots	Pathogens, organic material, nutrients
Septic systems	Pathogens, nutrients
Forests[c]	Organic material, nutrients, suspended sediment, toxics, water temperature increase
Rangelands[c]	Nutrients, suspended sediment, toxics

[a] *Can include untreated urban runoff during sewer overflows.*
[b] *Including construction sites.*
[c] *Atmospheric deposition contributes to nitrates and sulfates on these lands, especially in the East and Midwest.*
[d] *Mines can also yield point source pollution.*

cals, transportation by-products, construction-displaced sediment, and other wastes to rivers and streams. Farms are also important sources of all major categories of water pollution because of soil tillage, animal concentrations, fertilizers, pesticides, herbicides, and irrigation. Mines, landfills, animal feed lots, and rural septic systems are also common sources of some categories of pollution. Forests and rangelands are sources of nutrients and suspended sediments, and they can also yield toxics when pesticides and herbicides are used. Also, water temperature increases can occur in forest areas following harvest or fire.

The 1988 EPA Water Quality Inventory cited earlier, reports data on land area sources. Agricultural runoff was the most prevalent source, impairing 20% of the affected river miles and 17% of the affected lakes (table 4). Mining, municipal discharges, urban runoff, and stream channel modification, were other common sources. Silvicultural activities impaired 3% of the river miles and <1% of the impaired lake area. Effects of rangeland management were not included as a separate category. The National Fisheries Survey (Judy et al. 1984) also placed agricultural runoff at the top of the list, but ranked silvicultural activities second, affecting 8% of the river miles (table 4). These data have their weaknesses: States used a variety of methods to select which river and stream reaches to monitor for the EPA summary, and interpretations of water quality impairment varied too. But the data do provide a rough indication of the relative importance of silvicultural sources of water pollution.

Processes and activities potentially contributing to water pollution

Both human activities and natural processes can affect water quality (table 5). Human and animal waste production contribute to the accumulation of pathogenic organisms and organic material in water bodies. Forests and rangelands play a minor role here, contributing some decaying plant material and wastes of grazing livestock and wildlife.

Natural processes contributing nutrients include decay of organic material, nitrogen fixation, and dissolution of phosphorus-bearing rock. Major anthropic activities affecting nutrient concentrations are sewage and certain industrial effluents, urban runoff, concentrated livestock waste (e.g., at feedlots), use of synthetic detergents, agricultural use of fertilizers, and atmospheric deposition of air pollutants.

In a nationwide analysis based on stream nutrient sampling and land cover data for 928 watersheds, Omernik (1977) found that annual nutrient concentrations (both total phosphorus and total nitrogen) in agricultural watersheds were roughly 9 times higher than in forested watersheds and roughly 4 times higher

Table 4.—Incidence of major sources of water pollution in the United States according to two national surveys (% of assessed miles or acres).[a]

Source	Water quality inventory[b]		Fisheries survey[c]
	River and stream miles	Lake and reservoir acres	River and stream miles
Agricultural runoff	19.8	16.5	29.5
Municipal discharge	5.8	4.3	6.7
Resource extraction	4.7	1.2	3.3
Stream channel modification[d]	4.6	9.4	—
Storm sewers/urban runoff	3.2	7.8	4.2
Silviculture/forestry	3.1	0.3	7.5
Industry	3.0	2.2	4.9
Construction	2.3	0.9	3.1
Land disposal[e]	1.6	7.5	5.6
Combined sewers[f]	1.3	0.1	3.1

[a] *The same miles of river or acre of lake may be affected by more than one category of pollution.*
[b] *Source: EPA (1990). Indicates where "designated beneficial uses" were "not fully supported."*
[c] *Source: Judy et al. (1984). Indicates where fish "survival, productivity, or use" is "adversely affected." Also included were feedlots (5.0%), grazing (2.3%), and natural sources (22.2%).*
[d] *Channelization, dredging, dam construction, stream bank modification.*
[e] *Leachate from septic tanks, landfills, waste sites.*
[f] *Storm sewers and sanitary sewers that discharge untreated wastes during storms.*

Table 5.—Principal natural processes and management activities that may cause water pollution.

Category	Activity or process
Pathogenic organisms	Human and animal waste production
Organic material	Human and animal waste production Decay of natural plant material (e.g., from seasonal leaf fall) Slash production from harvesting Industrial effluents (e.g., pulp mills)
Nutrients	Atmospheric deposition Decay of organic plant material Fixation of nitrogen gas by plants Dissolution of phosphorus-bearing rock Urban waste release (sewage effluent) Runoff of agricultural fertilizers Livestock waste production Urban runoff Use of synthetic detergents (phosphorus) Release of nutrients following harvest or forest fire
Suspended sediment, silts	Bank erosion from floods Erosion following severe wildfire Erosion from agricultural practices Erosion from construction (e.g., land development) Erosion from timber harvest and road construction Erosion from overgrazing Mining
Dissolved solids	Dissolution of rock and soil Atmospheric deposition Urban sewage effluents and runoff Road salting Concentrating effects of irrigation
Toxics	Numerous municipal and industrial activities Vehicle use[a] Mining Pesticide and herbicide use
Miscellaneous	Temperature increases Use of water for cooling in thermal electric plants Harvest or fire damage of riparian vegetation Acidity Mine and tailings drainage

[a]*This cause has greatly diminished as leaded gasoline has been replaced.*

Table 6.—Mean concentrations and yields of plant nutrients for different land covers in the United States.[a]

	Concentration (mg/L)		Yield (kg/ha/yr)	
Land cover	Total N	Total P	Total N	Total P
≥ 90% forest	0.598	0.018	3.47	0.091
≥ 75% forest	0.643	0.024	3.54	0.129
≥ 75% rangeland	1.297	0.097	1.04	0.065
≥ 40% urban	1.818	0.092	7.30	0.347
≥ 75% agriculture	2.702	0.140	5.55	0.255
≥ 90% agriculture	5.354	0.161	9.54	0.266

[a]*Source: Omernik (1977).*

than in rangeland watersheds (table 6). The percent of the watershed in agricultural and urban uses correlated positively with nutrient concentrations in all regions of the United States, while percent of the watershed in forest cover was negatively correlated with nutrient concentrations in nearly all areas.

Omernik (1977) also found that yields per unit of surface area told a somewhat different story from concentrations. Yields from predominately agricultural watersheds were only about 3 times those of forests but from 4 (phosphorus) to 9 (nitrogen) times those of rangelands. The differences between concentrations and areal yields were partially due to differences in precipitation and runoff across the watersheds, with forests typically receiving more, and rangelands less, precipitation than farmland. In any case, at the regional level, forests and rangelands contribute relatively little nitrogen and phosphorus when compared with agricultural and urban land.

Suspended sediments result largely from erosion. Erosion is of course a natural result of precipitation and runoff. Principal human management activities that cause erosion are agricultural tillage, construction-related land disturbance, timber harvest and associated road construction, overstocking of livestock, and mining (table 5).

Resources for the Future estimated sediment discharge from nonfederal rural lands into rivers and streams in the United States (Gianessi et al. 1986). The analysis was based on the 1982 National Resource Inventory (SCS 1984) estimates of erosion at nearly 800,000 points nationwide, and on sediment transport and delivery predictions. For the nation as a whole, discharge rates from cropland were 5.2 times the rate from forestland (table 7). The rangeland rate was 1.5 times the forest rate. Of the total discharge from nonfederal lands, 57% was estimated to originate on cropland, 16% on rangeland, and 10% on forestland (table 7). Thus, forests and rangeland together were estimated to contribute one-fourth of the sediments reaching the nation's waters from private land.

Salts enter the nation's waterways from natural dissolution of rock and soil, from atmospheric precipitation (which contains ions from both natural and fossil fuel sources), and from activities such as municipal and industrial water treatment releases, de-icing of roads, and the concentrating effects of irrigation. Peters (1984) reported, based on analysis of the USGS's nationwide NASQAN data (Ficke and Hawkinson 1975), that the primary determinants of dissolved solids concentrations are rock type, precipitation quantity, precipitation quality, and to a lesser extent population density. His analysis did not look specifically at the effects of irrigation on evaporation and the consequent concentration of salts in return flow. In determining concentration of dissolved solids, Hem (1989) emphasizes the importance of rock type, precipitation quality, and human causes such as salting of roads and irrigation. Forest and range vegetation, or harvest and grazing activities, apparently do not add significantly to the salt content of the nation's rivers.[2]

Toxic chemicals, both natural and synthetic, are in wide use throughout the globe. Scores are used by modern consumers in industry, agriculture, and mining. Pesticides, including herbicides, are among the most common toxics in use in rural areas. Agricultural uses dominate, but pesticides are also used on forests and rangelands. Table 8 gives a rough indication of the relative application rates of pesticides in 1980 on agricultural and National Forest lands in the United States. Agricultural use per land unit was roughly 1000 times the forest use for insecticides, 600 times for herbicides, and 1300 times for fungicides. The short crop cycle in intensive farming, relative to silviculture, contributes to the much heavier use of such chemicals in agriculture. Most National Forest lands in the United States are not treated with such chemicals during a typical crop cycle (from 20 to 100+ yr), and lands that are treated seldom receive more than one application per cycle. Furthermore, because erosion rates are generally lower on forested land, less of the chemicals that are applied are transported to the streams.

Pesticide use rates vary considerably from one year to the next in response to the incidence of pest problems, so how representative is 1980 for National Forest land? As figure 1 indicates, total pesticide use varied on National Forests from 252 Mg (1 Mg = 1000 kg, or 2,200

[2] Timber harvest can also affect water quality through the temporary increases in runoff that follow harvest. Runoff increases can improve water quality by diluting concentrations of pollutants (see Brown et al. 1990 for an analysis of the benefits of diluting salts in the Colorado River Basin), or harm water quality if the runoff increase causes increases in stream bank erosion.

Table 7.—Sediment discharge from nonfederal rural lands into rivers and streams in the United States.[a]

	Rate (Mg/ha/yr)	Percent of total discharge
Cropland	0.94	57
Pasture	0.29	5
Rangelands	0.27	16
Forests	0.18	10
Other[b]	1.58	12

[a] Source: Gianessi et al. (1986).
[b] Farmsteads, mines, quarries, pits, etc.

Table 8.—Annual use of pesticides on agriculture and forestland.

	Total (Mg/yr)[a]		Per land unit (kg/km²/yr)	
	Agriculture	Forests[b]	Agriculture[c]	Forests[b,d]
Insecticides	138,924	71	90.3	0.092
Herbicides	202,030	169	131.4	0.220
Fungicides	22,700	9	14.8	0.011

[a]*For 1980, as presented by Norris et al. (1991, table 7).*
[b]*National Forest land only.*
[c]*Land area used for crops taken from ERS (1989).*
[d]*Land area taken from USDA Forest Service (1983).*

pounds) in 1980, to a maximum of 502 Mg in 1983 and 121 Mg in 1990. (High rates of insecticide use were associated with spruce budworm and grasshopper problems; high rates of herbicide use were associated with conifer release and site preparation emphases.) The pesticide application rate in 1980 was close to the 11-yr average depicted in figure 1. Only about 20% of the forests in the United States are in National Forests, and use rates on other forestland may be higher than the National Forest experience. Also, the types of pesticides used on forestlands have been changing over the past 20 yr toward less toxic chemicals (Norris et al. 1991).

Water Quality Trends

Two recent studies evaluated nationwide water quality data for trends. Smith et al. (1987) summarized trends for 1974-81, and Lettenmaier et al. (1991) did so for 1978-87. Both studies used the NASQAN data, the only nationwide water flow and quality data that were collected over several consecutive years using consistent sampling and analysis procedures.

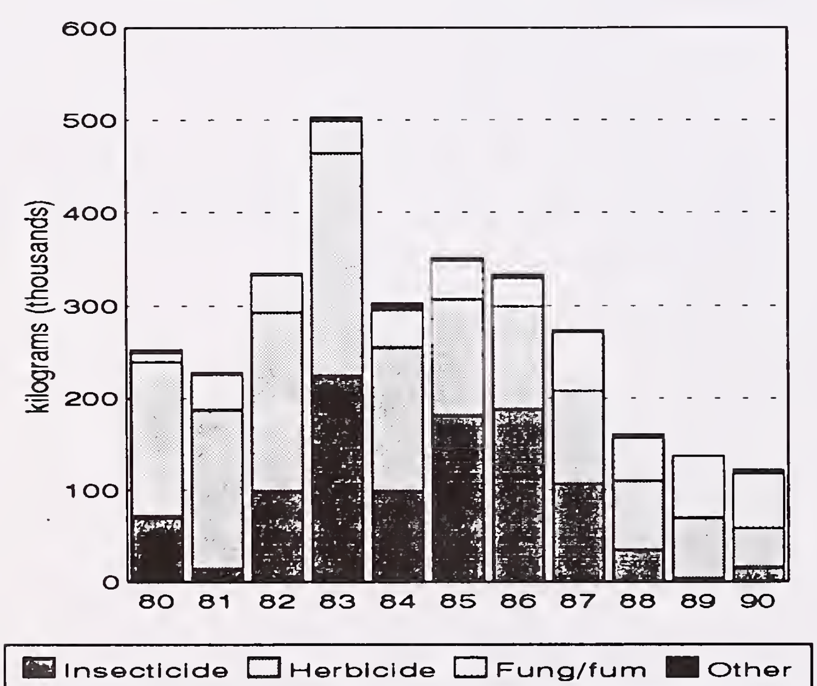


Figure 1. Pesticide use on National Forests 1980-1990. Source: USDA Forest Service. Report of the Forest Service (for fiscal years 1980-1990).

Lettenmaier et al. (1991) found that 15-20% of the stations had significant trends in pathogens and oxygen deficit (table 9). Stations with decreases were roughly twice as common as stations with increases. Increases in pathogens were associated with larger proportions of the land area in pasture or urban uses. Smith et al. (1987) found more decreases in fecal bacteria for the 1974-81 period (roughly 18% of the stations decreased while only 4% increased).

Both Smith et al. (1987) and Lettenmaier et al. (1991) found numerous stations showing increases in nitrogen and decreases in phosphorus concentrations. The latter study found that 21% of the stations increased in nitrogen concentration, while 6% decreased (table 9). The uptrends were distributed rather evenly over the continental United States. Smith et al. suggested that changes in agricultural fertilizer use and atmospheric deposition accounted for the significant trends, but an analysis by Lettenmaier et al. (1991) did not uncover significant associations for trend direction except for population density. For phosphorus, 18% of the stations showed decreases while 3% showed increases (table 9). The downturns occurred mainly in the Great Plains and in the East. Reasons for phosphorus trends were also unclear.

Lettenmaier et al. (1991) found few significant trends in suspended sediments; 12% of the stations showed decreases and 8% showed increases. No associations of trend with land cover were found. Earlier, Smith et al. (1987) reported that 14% of the stations decreased and 16% increased. Most of those increases occurred in the Columbia and Mississippi basins. Trends in suspended sediment concentrations were not significantly associated with total basin soil erosion rates, but increases in concentration were significantly related to (1) the fraction of total soil erosion contributed by cropland in the basin and (2) the absolute magnitude of cropland erosion in the basin. Trends were not associated with erosion rates on forests or rangelands, although the increases in the Columbia River Basin occurred mainly in areas with significant forest cover and timber harvest.

Both studies found many more increases than decreases in dissolved solids. Increases were most common in the eastern half of the country. Smith et al.

Table 9.—Water quality trends 1978-1987.[a]

Constituent	Number of stations	Percent of stations with significant trend[b]	
		Improving	Declining
Pathogens			
Fecal coliform	390	6	13
Fecal streptococcus	366	5	10
Organic material			
Oxygen deficit	316	4	12
Plant nutrients			
Total nitrogen	390	21	6
Total phosphorus	389	3	18
Suspended sediment	153	8	12
Dissolved solids			
Total dissolved solids	388	22	6
Toxics			
Arsenic	383	1	25
Cadmium	360	1	16
Lead	374	1	12
pH	378	24	3

[a]*Source: Lettenmaier (1991). Trends in flow-adjusted concentration.*
[b]*Significant at the 0.1 level.*

suggested the changes were caused by human waste discharges, salt use on roads, and surface coal production.

Lettenmaier et al. found that many stations showed decreases in toxics and increases in pH (table 9). The trace contaminant decreases were spread across the United States. Decreasing pH levels were most common in the East and Midwest, where atmospheric emissions are highest.

The NASQAN stations are generally located at the most downstream station in a hydrologic "accounting unit" (Ficke and Hawkinson 1975). Accounting units are relatively large drainage basins (there are 352 accounting units with a mean area of 26,638 km^2) that typically contain a variety of cover types and land uses, making it difficult to isolate the cause or source of the trends. Thus, neither study was able to offer definitive explanations for the changes. However, the explanations the studies gave tended to focus on agricultural, municipal, and industrial sources of pollution. Forest and rangeland areas were generally not mentioned, probably because they tend to yield relatively low concentrations of most constituents, which are difficult to isolate in the large watersheds.

The Following Chapters

The following chapters describe basic hydrology and water quality processes in forest and range watersheds. The chapters then summarize knowledge about impacts of forest and range practices on water quality. Of the long list of water quality parameters in table 1, the ones that present a major concern for forest and range watersheds are pathogens, temperature, dissolved oxygen, nitrate, and suspended sediment. Our discussion focuses on small watershed studies, many supported by the USDA Forest Service as part of the nationwide network of experimental watersheds. Detailed site summaries are provided at the end of each chapter.

Hydrology and Water Quality

The chemistry of water flowing from forests is typically very different from the chemistry of precipitation. Water quality depends in part on the patterns of precipitation quantity, the chemical composition of precipitation, and the interaction of water with plants, microbes, and soils. We begin this chapter with an overview of regional patterns in precipitation, and then we discuss the processes that contribute to changes in the chemistry of water as it flows through forests. We end by examining soil erosion and stream sedimentation. This chapter summarizes some aspects of water quality that apply across North America; many detailed examples are presented in later chapters.

Most of the forests in eastern North America receive more than 750 mm of precipitation annually (fig. 2) and many receive more than 1500 mm. Precipitation is generally well distributed through the year, with substantial amounts falling during the growing season. In contrast, many of the forests in western North America receive as little as 500 mm of precipitation annually, and the uneven distribution through the year results in considerable limitation on tree growth. Some portions, such as the Pacific Northwest, receive more than 2000 mm of precipitation.

A variety of processes contribute to the evaporation of water from forests (Brooks et al. 1991). Evaporation removes pure water, which concentrates dissolved chemicals in the water remaining. In eastern North America, evaporative processes remove about 600-750 mm, leaving about 150-1000 mm for runoff in streams. In the West, evaporative processes may remove about as much water as in the East, leaving only 150 mm of water for runoff. In the wettest portions of the Pacific Northwest, runoff generally exceeds 1000 mm (fig. 3).

These annual patterns span wide ranges of seasonal distributions of precipitation and runoff. At the Hubbard Brook Experimental Forest in New Hampshire, precipitation is evenly distributed through the year, but runoff shows a spring peak from snowmelt and a summer low from plant transpiration (fig. 4). The Fraser Experimental Forest in Colorado also shows relatively even precipitation through the year, with a spring peak in runoff driven by snowmelt. The H.J. Andrews Experimental Forest in Oregon experiences heavy precipitation in winter and drought in summer.

Water Quality Parameters

As noted in Chapter 1, water quality is commonly gauged by a range of physical, chemical, and biological parameters. Forest practices have the potential to alter physical and chemical characteristics of streams, which in turn have implications for biological features.

Physical parameters

The two most important physical characteristics of water quality are turbidity (or sediment concentration) and temperature. Another feature is the color of water imparted by dissolved organic molecules. Dissolved organic matter is generally unimportant in streams drain-

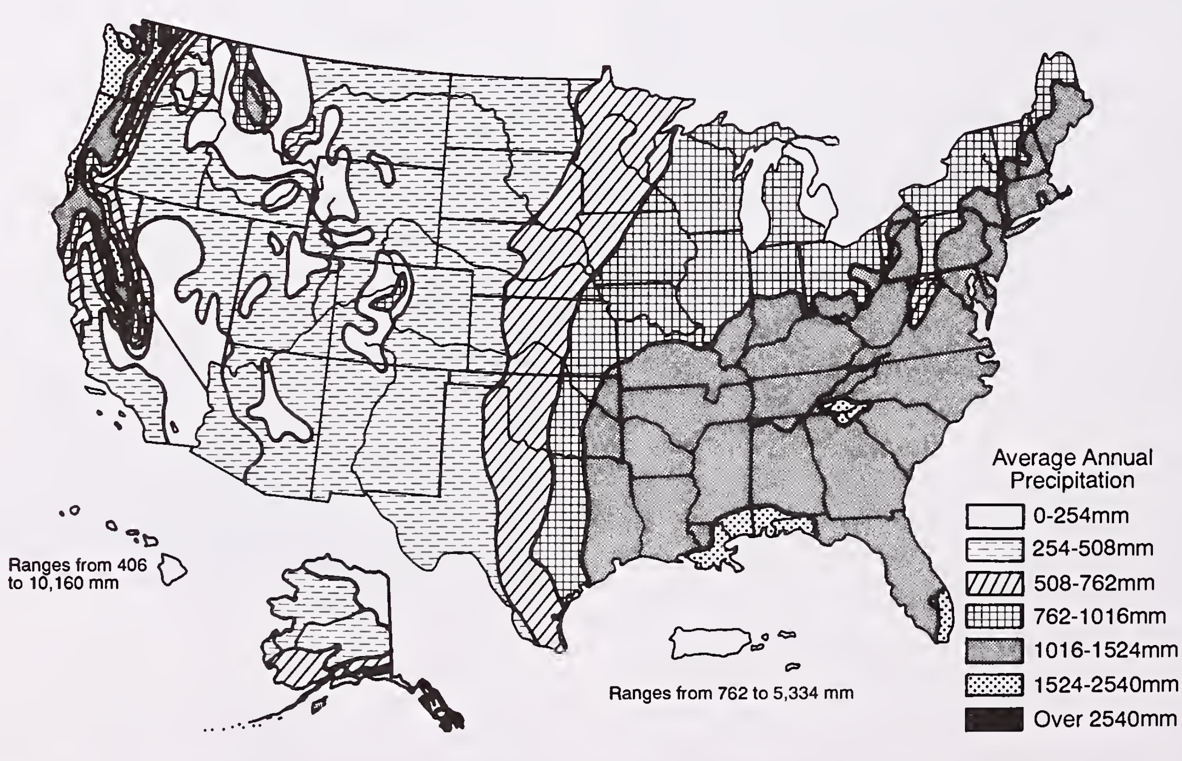


Figure 2. Average annual precipitation across the United States (USDA Forest Service 1982).

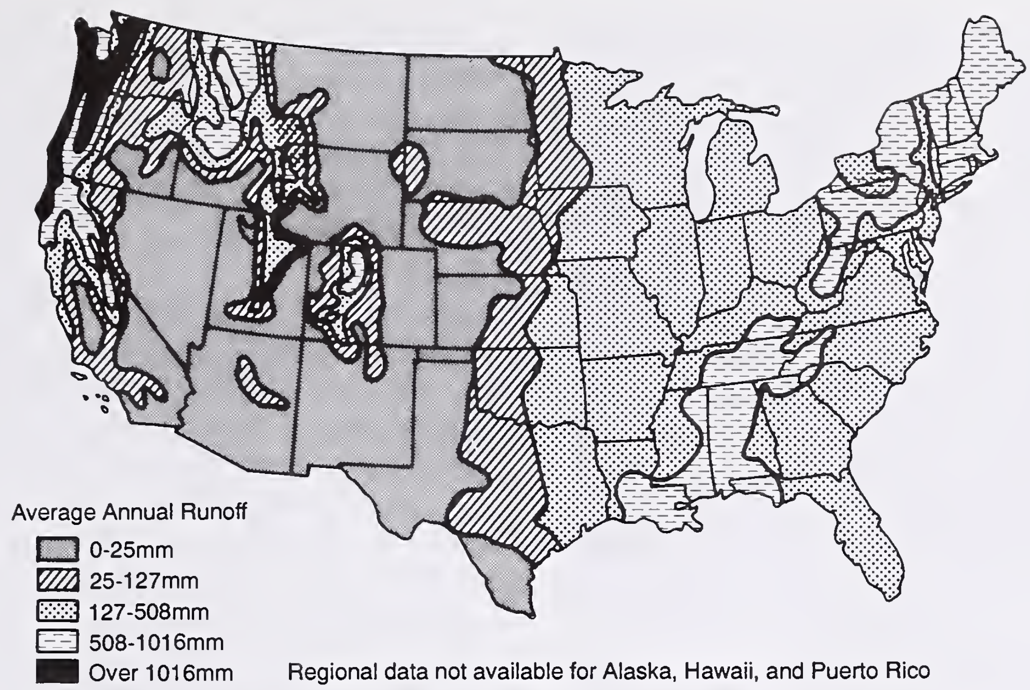


Figure 3. Average annual streamflow (runoff) across the United States (USDA Forest Service 1982).

ing forests, except for conditions in swamps or wetlands.

Turbidity is a measure of the particles suspended in water and is commonly gauged by a turbidimeter that measures the scattering of light passing through a water sample. Turbidity is reported in terms of the amount of light scattered (Nephelometric Turbidity Units [NTU]). Sediments are the suspended particles that scatter light, with the quantity measured as the weight of particles retained on a filter paper after the water has been filtered (in units of mg of sediment per L of water). Turbidity standards are typically included in water quality criteria because of ease of measurement in the field and the association between high turbidity and high sediment load (Brown 1989). Some turbidity standards are absolute, such as 1 NTU for municipal drinking water (MacDonald et al. 1991). Others specify relative increases above background levels, such as 5 NTU if background is less than 50 NTU, or a 10% increase if background levels exceed 50 NTU (MacDonald et al. 1991). Water quality standards may also include suspended sediment criteria, such as 500 mg/L for Oregon (Moore et al. 1979).

The relationship between turbidity and sediment content of the water is not constant; it varies with the size of particles and other factors. For example, the relationships between sediment concentrations and turbidity for 3 Oregon streams showed correlation coefficients (r^2) of 0.74 to 0.83 (fig. 5). The covariance between the water quality parameters is high but variable both within streams over time and especially between streams.

Turbidity and sediment concentrations of waters are important for several reasons. Turbidity by itself is unlikely to have major, immediate effects on fish; levels above 100,000 mg/L can be tolerated by many fish for short periods. However, sustained high turbidity can reduce photosynthesis by algae, reduce the success of sight-feeding fish, and perhaps alter food chains. Furthermore, fish generally require gravel beds for spawning; high sediment loads reduce the porosity of gravel beds, promote anaerobic conditions unsuitable for spawning, and block emergence of alevins from the gravels. Sediments are also a potentially important pathway for the removal of nutrients from watersheds, particularly phosphorus. Where sediment loads are extreme, siltation of rivers, lakes, and reservoirs may be a major problem.

Temperature affects both chemical and biological characteristics of streams. For example, the solubility of oxygen decreases rapidly as temperature increases. A change from 10 to 15 °C drops oxygen solubility by almost 20% (Golterman et al. 1978), and removal of tree canopies from over streams commonly raises monthly average stream temperatures by 3 to 7 °C or more (Brown 1989). Most aquatic organisms have optimal temperature ranges; forest practices that change temperatures more than about 2 °C from natural temperatures may be enough to alter development and success of fish populations (Hornbeck et al. 1984). Some fish species require a narrow range of temperature, whereas others are tolerant of wider ranges.

Light is not a physical characteristic of water quality per se, but it plays an important role in the effect of forest practices on aquatic ecosystems. Removing canopies that shade streams allows increased water temperature, increased photosynthesis, and a variety of other changes in the aquatic ecosystem.

Some of the most important effects of forest practices on aquatic ecosystems result from physical changes in the stream channel. Large woody debris in streams provide diversity of structure that helps maintain pools

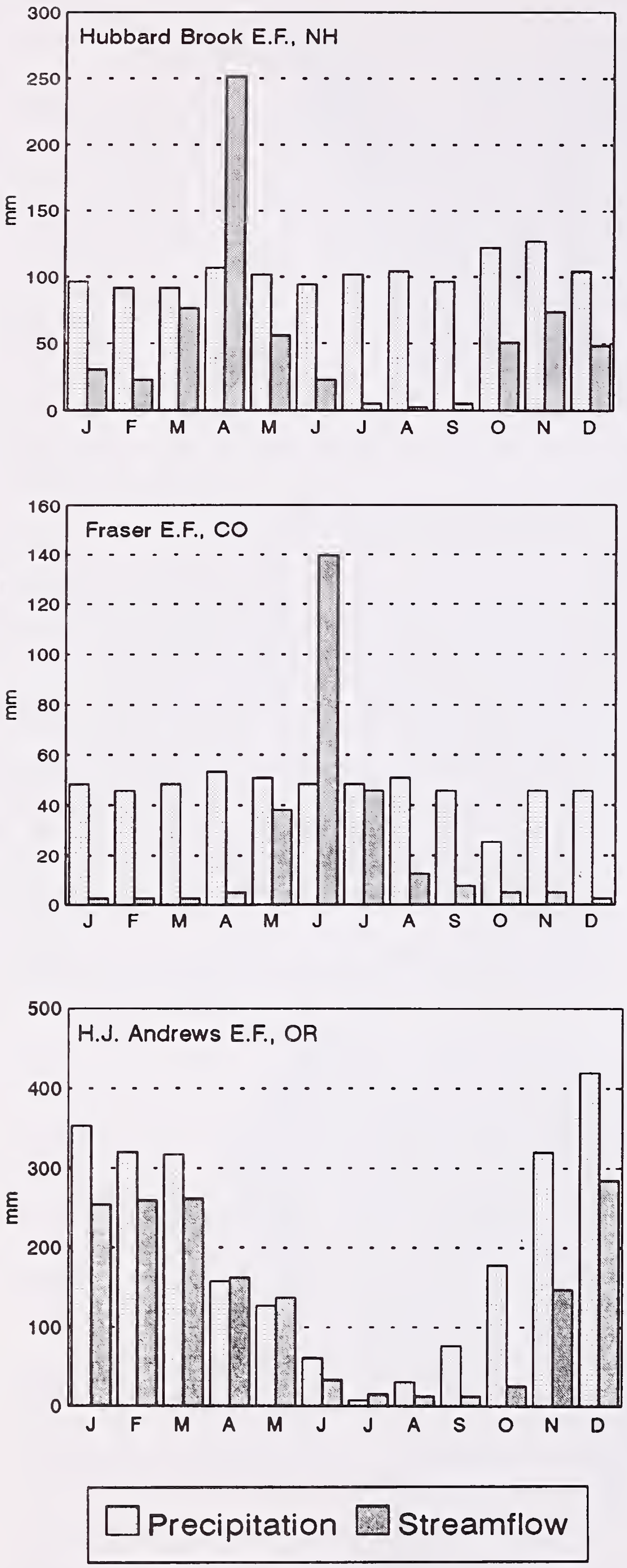


Figure 4. Annual patterns of precipitation and runoff for three USDA Forest Service Experimental Forests across the United States (Anderson et al. 1976).

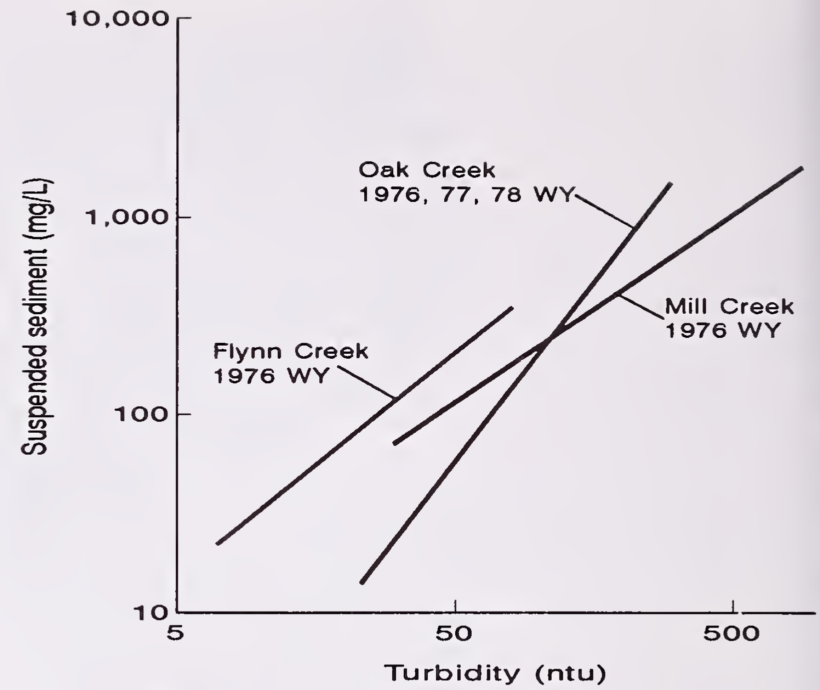


Figure 5. Relation between suspended sediment concentrations (Y-axis) and turbidity (X-axis) varies over time within sites (as indicated by $r^2 < 1$), and differs on average between sites (as indicated by different slope and intercept for lines for each stream) (Beschta 1980, cited in Brown 1989). For Flynn Creek, $S = 2.35T^{1.14}$, where S = suspended sediment (mg/L) and T = turbidity (ntu), $r^2 = 0.83$, and n = 86. For Mill Creek, $2.60T^{0.97}$, $r^2 = 0.74$, and n = 29. For Oak Creek, $S = 0.06T^{1.73}$, $r^2 = 0.78$, and n = 247.

and diverse aquatic ecosystems (Sedell et al. 1988). Harvest activities commonly have direct and indirect effects on woody debris in streams. Old-growth forests in the Pacific Northwest typically contain about 20 to 30 pieces of woody debris in every 100 m reach, compared with just 1 to 5 pieces/100 m in second growth forests (Sedell et al. 1988). In the 1950's and 1960's, woody debris was actively removed from streams as a part of harvesting operations to prevent depletion of stream water oxygen and to reduce obstructions to fish migration. Woody debris is now recognized as a beneficial feature in stream ecosystems. Although best management practices in the Pacific Northwest generally include provisions for retention of large woody debris (MacDonald et al. 1991), the implementation of the provisions is not always satisfactory (R. Beschta, Oregon State University, personal communication).

Two other physical aspects of stream channels are strongly influenced by forest practices: content of fine particles in gravel beds, and width and depth of channels (MacDonald et al. 1991). Relatively slight increases in the content of fine particles in gravel beds can substantially lower permeability and successful spawning and emergence by salmonid fish (see Chapter 7). Forest harvest without retention of streamside vegetation buffers can result in drastic reductions in stream depth and increases in stream width (see Chapter 7). For example, the width of Carnation Creek increased by more than 8 m in just 6 yr after logging the streamside forest (Scrivener 1988). Increased flows from harvested forests may also contribute to changes in channel morphology. These features are not generally gauged as water quality parameters, but best management practices (particularly within riparian areas) designed to sustain high quality of water may protect these important features as well.

Chemical parameters

Water chemistry involves the concentrations and interactions of many chemicals. The concentrations of these chemicals depend in part on biological interactions with aquatic plants, microorganisms, and animals. Of the wide array of chemicals present in streams, water quality concerns focus on just a few chemicals or groups of chemicals: nitrate, phosphate, specific conductance, dissolved oxygen, acidity, aluminum, heavy metals, and organic pollutants. Of these parameters, forest practices may have important effects on concentrations of nitrogen (particularly nitrate), dissolved oxygen, and perhaps phosphate; impacts on the other parameters are generally much less important (MacDonald et al. 1991).

The quality of water draining intact forested watersheds is typically very high. The nitrate concentration of streams draining forested areas averages about 0.23 mg/L in the United States, compared with an average of 3.2 mg/L for agricultural lands (Omernik 1976). The proportional difference for phosphate concentrations is similar: 0.006 mg/L for forestlands, compared with 0.06 mg/L for agricultural lands. Water quality concerns on forestlands focus strongly on the changes that result from management activities, such as increased concentrations of nitrate following harvesting.

Forest practices may elevate the concentrations of many chemicals in stream water, but only two are of significant concern. Nitrate-nitrogen concentrations above 10 mg/L are unacceptable for drinking water because of risks to infants. Phosphate concentrations do not reach levels of concern for drinking water, but a standard of 0.1 µg/L has been set to prevent eutrophication of estuaries (MacDonald et al. 1991). No standard has been set for freshwaters because risks of eutrophication are very location dependent; a level of 0.05 mg/L may be sufficient to protect lakes (MacDonald et al. 1991). A variety of forest practices usually result in elevated concentrations of nitrate in streams, but concentrations are usually well within drinking water standards (important exceptions do occur). Forest practices typically have little (if any) effect on phosphorus concentrations (Salminen and Beschta 1991).

Forest practices may also alter the acidity of stream water, both in terms of the actual concentrations of H^+ (typically gauged as pH, the negative of the logarithm of the H^+ activity) and the ability of the water to buffer inputs of H^+ (the acid neutralizing capacity, in terms of µmol of H^+ that could be consumed per L of water). The most extreme example of changes in stream acidity come from the devegetation experiment at the Hubbard Brook Experimental Forest (Likens et al. 1970). All trees were cut and left in place in a watershed, and revegetation was inhibited by heavy use of herbicides. This drastic treatment resulted in a decline in stream pH from 5.1 to 4.3, and the ability to buffer H^+ dropped from about 20 µmol/L to 0. The concentration of aluminum also increased 10-fold, from about 0.2 mg/L to 2.0 mg/L, with potential adverse effects on aquatic organisms.

As noted above, the concentration of oxygen dissolved in stream water is important for fish and other components of aquatic ecosystems. Streams typically contain about 5 to 10 mg of oxygen/L, with lower concentrations for streams with high organic matter and high temperature. For example, Hall and Lantz (1969; cited in Brown 1989) found that concentrations of oxygen in a stream flowing through a clearcut dropped to as low as one-third of the concentration found in a similar stream draining an intact forest. Streams containing spawning salmonid fish should not drop below 8 mg of O_2/L for 1 day, or below 9.5 mg/L for a 7-day mean; concentrations of 5 to 6.5 mg/L may be sufficient for adults (MacDonald et al. 1991).

Specific conductance (or more properly, the electrolytic conductivity) is a parameter of the chemical status of a stream that is widely measured. Specific conductance simply represents the ability of water to conduct an electric current, which depends upon the total quantity of dissolved ions which ions are present. Common background levels for forest streams fall in the range of 3 to 15 milliSiemens (mS) per meter of water. Specific conductance is easily measured with a small, portable meter and may provide a good overall indication of whether forest practices have altered water chemistry.

Microbiological parameters

Most interest on microbiological quality of water centers on organisms that are pathological to humans, or on generalist bacteria (such as total coliform counts) that may be overall indicators of microbial contamination (MacDonald et al. 1991). Total coliform counts have been widely used to assess drinking water quality, with a standard count of 0 to 1 colony per 100 mL as the acceptable level. Two more specific classes of bacteria are commonly examined: fecal coliform (largely from feces of humans and other mammals) and fecal streptococcus (mostly from mammals other than humans). The ratio of fecal coliform to fecal streptococci may be useful to differentiate between human and animal sources of pollution.

Giardiasis, a waterborne disease caused by *Giardia lamblia* (a flagellated protozoan), is a major water quality concern in many western mountains (Brown 1989).

Many cases of giardiasis can be traced to streams with substantial beaver activity, but the role of other mammals in spreading the disease is not clear.

Grazing is the primary land use practice that may increase microbial contamination in forest streams, although concentrated recreation use and wildlife populations could increase levels to unacceptable concentrations in some cases. In a watershed-level study in the Bear River Range of northern Utah, Darling and Coltharp (1973, cited in Brown 1989) found maximum total coliform counts of about 150 colonies/100 mL in an ungrazed watershed, compared with maximums of 700 colonies/100 mL for a sheep-grazed watershed and 1500 colonies/100 mL for a cattle-grazed watershed.

Factors That Change Water Chemistry

The chemistry of water flowing through forests changes as water passes through the canopy, soil, and subsoil into streams. Water chemistry profiles characterize these changes, and the changes can be substantial (fig. 6). For example, throughfall beneath the canopy of a young red alder/Douglas fir stand in British Columbia was enriched in sulfate, chloride, and alkalinity (acid neutralizing capacity) relative to precipitation, but nitrate was depleted (Binkley et al. 1982). Water percolating through the forest floor picked up large quantities of nitrate, but much of the nitrate was removed before the water reached the stream.

Canopy interactions

Precipitation chemistry changes substantially as water passes through forest canopies. Water passing through canopies is designated as throughfall, and water running down tree trunks is stemflow. The proportion of precipitation that is lost from the canopy through evaporation (interception loss), and consequently the portion becoming throughfall and stemflow, depends on precipitation and stand characteristics. In large rainfall events, nearly all the precipitation becomes throughfall and stemflow. Rainfall events of 2-3 mm or less may not exceed the water storage capacity of canopy surfaces and result in no throughfall or stemflow. Stands with high canopy surface areas have greater storage capacity for water and therefore greater opportunity for interception losses. In general, throughfall typically ranges from 60 to 95% of rainfall, with stemflow accounting for 0 to 35% (but typically < 10%). Interception losses remove about 5 to 35% of precipitation (Parker 1990). Forest harvesting reduces interception losses, allowing greater quantities of water to reach the soil.

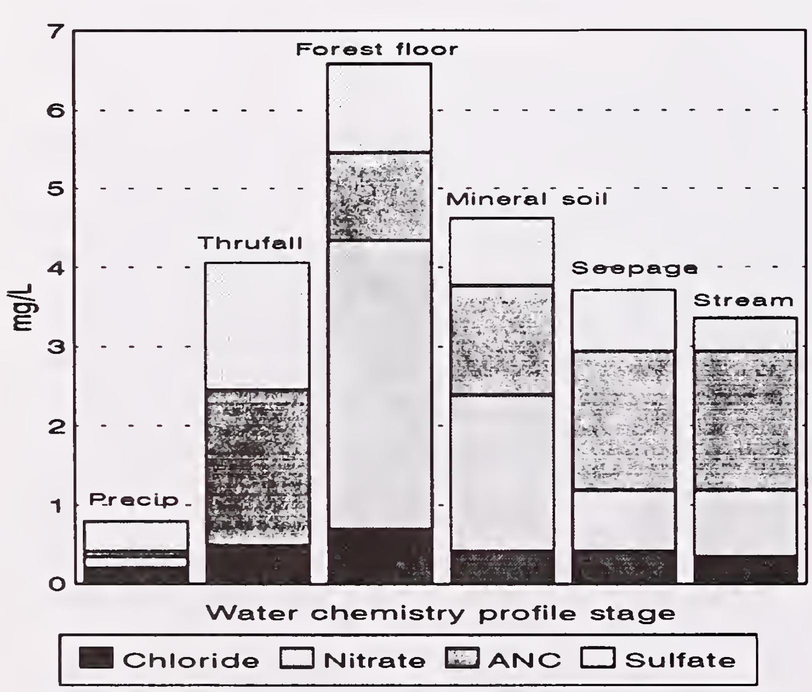


Figure 6. Changes in anion chemistry in water passing through a young red alder/Douglas fir forest in British Columbia (from data of Binkley et al. 1982).

Key processes that determine the differences in chemistry of incoming precipitation, throughfall, and stemflow are

1) concentration of chemicals in precipitation from interception loss of water;
2) wash-off of chemicals deposited on canopy surfaces since previous precipitation events;
3) wash-out of chemicals that were previously within the leaves;
4) chemical adsorption onto exchange sites in the canopy; and
5) uptake of chemicals into the leaves, microbes, lichens, and other epiphytes in the canopy.

If all else were equal, a 20% interception loss would increase ion concentrations in throughfall by 25%. (A constant quantity of ions dissolved in only 80% of the original volume of water gives a 25% increase in concentration.) Parker (1990) summarized available information on the total effect of these throughfall processes and found that in almost all cases the concentrations of ions are increased in throughfall relative to precipitation (fig. 7); only the ammonium and nitrate forms of nitrogen were occasionally depleted in throughfall.

Further processing of throughfall solutions by understory canopies can substantially alter the chemistry of precipitation reaching the forest floor surface. For example, Yarie (1980) compared throughfall chemistry above and below the understory canopy of 3 types of forests in coastal British Columbia and found substantial reductions in concentrations of nitrate, ammonium, and phosphate.

Interception losses are also substantial from snow sitting in canopies; Meiman (1987) noted that about 30 - 35% of incident snowfall may be lost through interception in Rocky Mountain forests. Most snow falls to the ground before melting occurs, so opportunities for

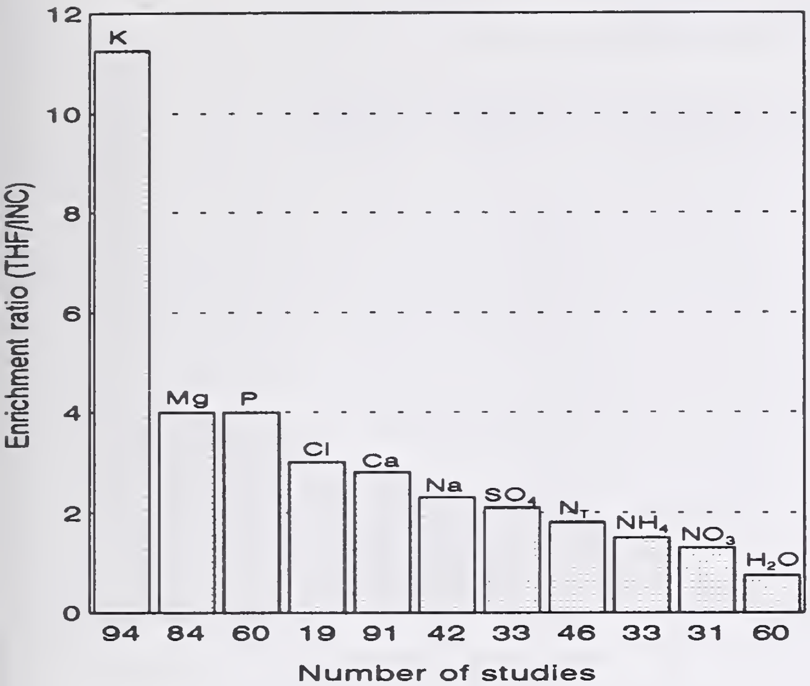


Figure 7. Ratio of concentrations of elements in throughfall (THF) to precipitation (INC) from a literature summary by Parker (1990). The number below the X-axis indicates the number of studies that went into the comparison for each element.

changes in the chemistry of "throughfall" are less than canopy interactions with rainfall.

Forest floor interactions

The forest floor is the first horizon of most forest soils, consisting mostly of organic materials in various stages of decay. Key processes that alter water chemistry are: removals of chemicals by microbial (or root) uptake and ion exchange, and release of chemicals from microbial decomposition and ion exchange. Evaporation of water from the forest floor could also serve to concentrate forest floor solutions. Comparisons of chemistry between throughfall and forest floor leachates show no consistent pattern within vegetation types or within individual elements (table 10).

Forest burning and harvest can result in forest floor disturbance that allows precipitation to reach mineral soil horizons unaltered. Fire increases the pool of available nutrients at the same time that uptake by plants may be reduced. The effects of light surface fires are generally slight, whereas wildfires that consume forest canopies may have major effects. Richter et al. (1982) examined the response of stream chemistry to prescribed fires in loblolly pine forests and found no effects. Tiedemann et al. (1978) examined the effects of a severe wildfire in the Entiat Experimental Forest in Washington. After the fire in the Douglas fir forest, peak nitrate-nitrogen levels rose from normal levels of 0.02 mg/L to 0.6 mg/L, still well below any water-quality threshold of concern. Leaching losses of nutrients after fires are generally low (Beschta 1990).

Mineral soil interactions

A wide range of processes alter the chemistry of water moving through mineral soils, including

1) uptake of water and chemicals by tree roots, mycorrhizae, and microbes;
2) release of chemicals from decomposition, including oxidation of N and S compounds;
3) exchange reactions that release and adsorb chemicals, reversibly or irreversibly;
4) mineral weathering (and perhaps secondary mineral formation); and
5) chemical precipitation (formation of low-solubility salts) and dissolution.

The combination of these processes generates a wide range of patterns of change in chemistry as water percolates from the forest floor through to the B horizon of the mineral soil (table 11). The only generalization apparent across species or elements is that forest floor leachates

Table 10.—Ratio of chemical concentrations in throughfall to forest floor leachates from the Integrated Forest Study (calculated from Johnson and Lindberg 1992). Ratios less than 1 indicate greater concentration in forest floor leachate than in throughfall.

Site	Nitrate	Sulfate	Ammonium	Potassium	Calcium
Northern Hardwoods					
Ontario	0.6	1.3	1.1	1.3	0.3
New York	0.7	0.8	3.1	0.8	0.2
Douglas fir	16.0	1.3	1.8	15.3	0.1
Red alder	0.01	1.5	0.3	0.3	0.1
Pacific silver fir	9.5	2.1	2.4	0.9	0.7
Loblolly pine					
Tennessee	3.1	1.1	30	0.4	0.4
N. Carolina	59.0	1.9	11	1.0	0.9
White pine	1.2	0.6	0.5	0.4	0.2
Red spruce/balsam fir	1.4	0.6	15	11	0.6

Table 11.—Ratio of chemical concentrations in forest floor leachate to B horizon leachate from the Integrated Forest Study (calculated from Johnson and Lindberg 1991). Ratios less than 1 indicate greater concentration in B horizon leachate than in forest floor leachate.

Site	Nitrate	Sulfate	Ammonium	Potassium	Calcium	Aluminum
Northern hardwoods						
Ontario	0.3	0.8	5.6	10	0.7	—
New York	4.3	1.1	7.8	15	2.6	1.2
Douglas fir	4.0	0.7	6.0	6.6	6.8	216.0
Red alder	1.8	2.6	28	5.1	1.5	20.0
Pacific silver fir	0.7	0.7	2.0	6.6	1.5	2.4
Loblolly pine						
Tennessee	6.2	0.6	2.0	5.9	1.1	22.0
N. Carolina	7.0	0.3	0.8	1.1	0.7	165.0
White pine	147.0	5.6	28	17	31	47.0
Red spruce/balsam fir	1.2	1.5	1.7	0.6	1.5	—

are typically more concentrated than B horizon leachates, though exceptions to this pattern are common.

Harvesting reduces nutrient uptake and may also increase soil decomposition rates, increasing the pool of available nutrients in the mineral soil. A wide range of studies have examined the response of stream chemistry to harvesting (detailed in later chapters). The range of responses is bracketed by comparing two forests: an old-growth forest of Douglas fir in the H.J. Andrews Forest in Oregon, and a second-growth forest of northern hardwoods in the Hubbard Brook Experimental Forest in New Hampshire. The forests had similar total nitrogen contents of about 5,000 kg/ha. Stream concentrations of nitrate-nitrogen rose following harvesting from the precutting level of 0.006 mg/L to 0.1 mg/L at the Oregon site, and from 0.5 mg/L to 10 mg/L at the New Hampshire site (Sollins et al. 1980). The major difference controlling the responses of elements to harvesting was likely the carbon content of the soils. The Oregon site was very rich in carbon (about 1300 Mg/ha) compared to the New Hampshire site (about 340 Mg/ha), and greater carbon availability may allow soil microbes to immobilize more N into soil humus.

In-stream processes

The chemical and physical characteristics of stream water also depend on processes occurring within the stream. Periphyton (algae attached to rocks or other surfaces in the stream) and microbes take up nutrients, and decomposition processes release nutrients. Major oxidation and reduction reactions occur, which may increase or decrease nitrate concentrations. One of the best examples of the magnitude of control that in-stream processes play was demonstrated by removing debris dams from a 175-m stretch of a second-order stream in the Hubbard Brook Experimental Forest (Bilby and Likens 1980). The export of fine particulate organic matter increased 6-fold, and output of larger organic matter (> 1 mm) more than doubled.

Fertilization and herbicide application

Fertilization adds very large pulses of nutrients that may exceed the immediate uptake ability of trees. In general, careful fertilization does not increase stream water concentrations of nitrate-nitrogen to potentially toxic levels (Fredriksen et al. 1975; Miller and Fight 1979; Hetherington 1985; Norris et al. 1991).

Three aspects of herbicide application to forests may influence water quality: concentrations of herbicides in streams, response of stream chemistry to herbicide treatment, and effect of treatment on erosion. Herbicides may enter streams by direct application or by movement from the soil (either on soil particles or dissolved in water). Fredriksen et al. (1975) summarized a range of studies with several herbicides and concluded that concentrations in streams were too low to warrant concern (peaking at about 0.01 mg/L within hours of application, declining to < 0.001 mg/L after weeks). No reports have appeared of injury to stream biota from herbicide applications that followed regulatory guidelines (Newton and Norgren 1977; Norris et al. 1991).

Almost no information is available on the effects of herbicide treatments on the concentrations of elements in stream water. Bigley and Kimmins (1983) treated a 9-yr-old Douglas fir plantation in British Columbia with glyphosate and found that nitrate-N concentrations in soil leachates climbed from about 0.8 mg/L to 2.5 mg/L after 3 months. Miller (1974) used phenoxy herbicides to remove nitrogen-fixing red alder from stands with conifers; nitrate-N concentrations were relatively high (1-2 mg/L) in both control and treated forests.

The indirect effects of herbicide treatments on soil erosion have received little attention. A recent book on vegetation management for conifer production (including a strong emphasis on herbicides) does not list erosion in its index (Walstad and Kuch 1987). Applications to well-established forests probably have negligible effects. Herbicides are also used (in combination with other treatments) to establish pine plantations in the Southeast, and the herbicides may substantially increase soil erosion. Pye and Vitousek (1985) found that extremely intensive site preparation (harvesting, shearing stumps, windrowing slash, and disking soil) resulted in about 4 Mg/ha of erosion in the first year; adding a herbicide treatment to inhibit hardwoods raised erosion to 10 Mg/ha.

Stream sedimentation

Movement of soil particles into streams may be the most visible connection between forests and streams. Forests comprise about one-third of the contiguous United States, and rangelands account for another one-third. Forests also tend to occupy the steepest and therefore most erosion-prone portions of landscapes. However, the two-thirds of the country occupied by forests and rangelands contribute only about one-fourth of the total sediment discharge in the country (Gianessi et al. 1986). Nonetheless, about half of the major forested drainage basins in the contiguous United States were classified as "affected by pollution from silvicultural activity" by the U.S. Water Resources Council (1978).

The delivery of sediment to streams can involve a variety of processes. Large quantities of sediments can be delivered directly by mass movement of blocks of soil. More moderate quantities can be derived from soil particles that are loosened by the impact of raindrops and transported in overland flow. In most cases, sediment-laden overland flow travels only short distances if forest floor materials are relatively intact on the soil surface. The energy available to loosen and carry sediment depends on the size of raindrops and their velocity. At a rate of 1 mm of rainfall/hour, the total energy available to detach soil particles is on the order of 15 Joules/mm of rain (Miller 1977). Intensities of greater than 10 mm/hr have 25 to 30 Joules of energy for each mm of rain, resulting from larger size and velocity of raindrops. The intensity of throughfall under forest canopies is not very sensitive to rainfall intensity; when the intensity is sufficient for throughfall to occur, drop size tends to be large, and canopy height of 10 m or more allow drops to attain near-maximum velocities (Miller 1977, USEPA 1980). Understory canopies may reduce the energy of raindrops, and the presence of forest floor horizons reduces energies to near zero.

Fire may increase soil erosion and stream sedimentation (see Kittredge 1948 for a good review). Fire may remove much or all of the forest floor, exposing mineral soil to the impact of raindrops. The mineral soil may also have reduced infiltration capacity that results from dispersion of soil aggregates by raindrop impacts and from fire-induced hydrophobicity (Pritchett and Fisher 1987). The amount of increased erosion after fire depends heavily on steepness of slope, fire intensity, plant recovery, and rainfall intensity. In general, the effects of fire on erosion of forest soils is relatively minor and declines rapidly as revegetation proceeds (Striffler and Mogren 1971). In some cases, however, the effects can be dramatic, particularly on steep slopes (Beschta 1990).

Slash fires after logging can also increase erosion rates. Fredriksen (1970) found that slash burning after logging a forest in Oregon increased stream sediment loads by more than 30 times the sediment loads of cut but unburned areas.

The forest practices with the greatest potential for causing erosion and stream sedimentation are road construction and intensive site preparation. In the Pacific Northwest, Fredriksen (1970) estimated that road cutbanks and fill slopes lost about 1 cm/yr over a 5-yr period, and that the roads lost more than 7 cm/yr. In the Southeast, Hewlett (1982) examined erosion following harvesting of a pine forest in the Georgia piedmont. Annual sediment delivery to the stream increased from about 0.01 Mg/ha before logging to 4 Mg/ha after logging; about 90% of the increase resulted from poor road location and maintenance and from direct damage to the stream channel by logging equipment. In North Carolina, Pye and Vitousek (1985) reported that although moderately intense site preparation treatments (roller chopping and burning slash) had little effect on erosion after harvest, soil disking and herbicide application had major effects (as noted earlier). Patric et al. (1984) estimated that erosion rates are less than 0.2 Mg/ha for about one-third of the forests in the eastern and interior West forests, and less than 0.25 Mg/ha for three-fourths of these forests. Few forested watersheds show erosion rates of more than 1 Mg/ha annually. Burger (1983) estimated likely erosion rates for flat sites in the Southeast (table 12). Natural rates range from near 0 to about 0.05 Mg/ha. Typical logging (including road construction) increases the rate to 0.1 - 0.5 Mg/ha. More intensive disturbance of the site (such as windrowing slash or disking the soil) can raise rates up to almost 10 Mg/ha annually for a few years. While most erosion studies assess the mass of soil that moves downslope, the amount that actually makes it to the stream may be a small fraction of the total (5-10%) (Hewlett 1982).

Poor management of livestock grazing greatly accelerates erosion. A classic study in the Wasatch Range in

Table 12.—Erosion rates for site preparation treatments following forest harvest in the Southeast United States (Burger 1983).

Treatment	Recovery time (years)	Annual erosion rate (Mg/ha)
Natural	—	0 to 0.05
Logged, with roads	3	0.1 to 0.5
Burned	2	0.05 to 0.7
Chopped	3	0.05 to 0.25
Chopped and burned	4	0.15 to 0.40
Windrowed	4	0.2 to 0.24
Disked	4	2.5 to 10

Utah showed that overgrazing by sheep increased soil erosion from about 0.1 Mg/ha to over 15 Mg/ha (Noble 1965, Brown 1989). Increased erosion from heavily grazed lands is from increased energy of raindrops that fall directly on soil; reduced trapping of mobilized sediments by plants and plant debris; and reduced infiltration rates that result from soil compaction (Moore et al. 1979; Gifford and Springer 1980). Although many studies have characterized grazing impacts on vegetation, soil physical properties, and small-scale erosion, few studies have directly examined the connections between grazing impacts and water quality parameters.

Mass movements. Many forest soils occur on steep slopes that are prone to large soil mass movements. Classification of mass movements include debris slides, debris avalanches, debris flows, and debris torrents, all involving the initial failure of a shallow, cohesionless soil mass on steep slopes (Swanston 1974; Swanston and Swanson 1976; Swanston 1991). Debris slides are relatively dry compared with wetter debris avalanches and water-saturated debris flows. Debris torrents flow down stream channels rather than slide down hillslopes. Debris avalanches tend to move about 10 to 100 m^3 of material per km^2 of forest in the high precipitation environments of the Pacific Northwest (Swanston and Swanson 1976).

Another class of mass movements includes soil creep, slumps, and earthflows. Creep is the gradual downhill movement of cohesive soil; rates of about 1 m in 100 yr are common for creeping slopes (Swanston and Swanson 1976). Slumps involve a rotation of a block of soil, giving a spoon-shaped headwall and some lateral movement of the block. Earthflows are larger and involve a series of slumps and flows down a slope, with typical rates of movement of about 10-25 m in 100 yr.

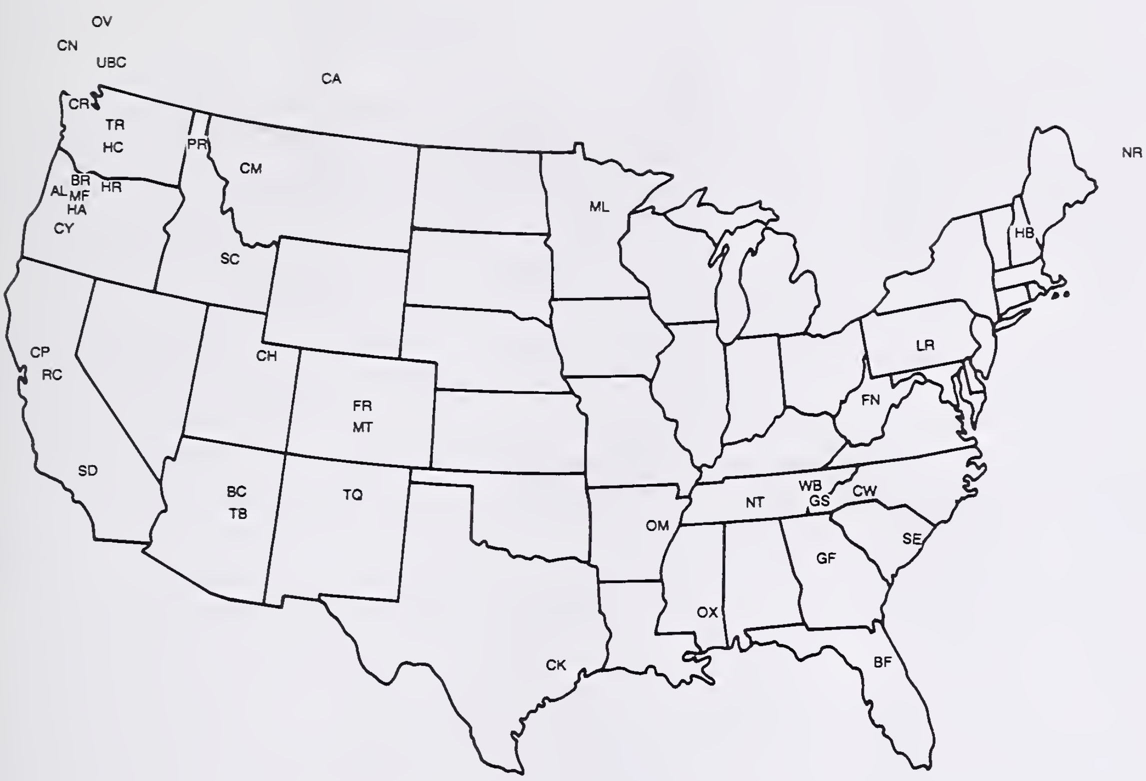
Forest practices, especially road construction, may substantially increase the incidence of mass movements in steep terrain. Harvesting typically leads to greater soil moisture content because of reduced water loss through interception and transpiration, and wet soils are weaker than drier soils. The decay of roots that contribute substantially to the strength of soils may also allow more slope failures. Road construction enhances risks of slope failures by collecting and concentrating water moving downslope and by increasing slope angles on the cut (upslope) and fill (downslope) sides (Swanston and Swanson 1976). In the H.J. Andrews Experimental Forest, Swanson and Dyrness (1975) evaluated 25 yr of debris flow patterns. Forested slopes accounted for about 35 m^3 of material per km^2 annually, compared with 130 m^3/km^2 for clearcut slopes and 1800 m^3/km^2 for roaded areas.

In relation to water quality, mass failures of stream banks may be particularly important. The failure of an undercut bank, or degradation of streamside vegetation by overgrazing, may have much larger impacts on stream sedimentation than activities dispersed throughout the watershed.

Sources of Water Quality Data

The summaries of the effects of management practices on water quality presented in later chapters come from two general types of sources: the USDA Forest Service's watersheds on experimental forests and the U.S. Geological Survey's large "benchmark" watersheds. A wealth of site-specific information is available from intensive studies of individual, small watersheds. The USDA Forest Service has monitored the quality of water draining forested watersheds at experimental forests across the United States and gathered information on both background conditions and responses to major management activities. Other groups in the United States and Canada have conducted similar small-watershed studies. These detailed studies form the foundation of our assessment of the effects of forest practices on water quality. Their locations are shown in figure 8. We also supplement these watershed-level examples with information from (1) smaller-scale studies that help provide information on mechanisms by which management practices affect water quality, and (2) broader-scale studies that have assessed regional impacts of forest practices.

The U.S. Geological Survey also monitors water quality parameters at many sites. Among those sites is a set of benchmark stream gauging stations, which monitor water quality of watersheds that are little affected by human actions (Biesecker and Leifeste 1975). Of the approximately 55 benchmark basins, we have



Key to study locations:

AL	Alsea Watersheds
BC	Beaver Creek Experimental Watersheds
BF	Bradford County
BR	Bull Run Watersheds
CA	West Central Alberta
CH	Chicken Creek
CK	Cherokee County
CM	Coram Experimental Forest
CN	Carnation Creek
CP	Caspar Creek Experimental Forest
CR	Clearwater River
CW	Coweeta Hydrologic Laboratory
CY	Coyote Creek Experiemental Forest
FN	Fernow Experimental Forest
FR	Fraser Experimental Forest
GF	Grant Memorial Forest
GS	Great Smoky Mountains National Park
HA	H.J. Andrews Experimental Forest
HB	Hubbard Brook Experimental Forest
HC	Hansel Creek
HR	High Ridge Watersheds
LR	Leading Ridge
MF	Middle Fork Santiam River
ML	Marcell Experimental Forest
MT	Manitou Springs Experimental Forest
NR	Nashwaak River
NT	Natchez Trace State Park
OM	Ouachita Mountains
OV	Okanagan Valley
OX	Oxford, Mississippi
PR	Priest River Experimental Forest
RC	Redwood Creek
SC	Silver Creek
SD	San Dimas Experimental Forest
SE	Santee Experimental Forest
TB	Three Bar Experimental Watersheds
TQ	Thompson Research Center
TR	Tesuque Watersheds
UBC	University of British Columbia
WB	Walker Branch Watershed

Figure 8. Map of key study locations.

selected 43 (in 33 states) that are largely covered with forest or rangeland vegetation. (See table A.1, in the appendix, where the stations are grouped by region to correspond with the regional chapters that follow.) These 43 watersheds average 228 km^2 in surface area, which is considerably larger than a typical experimental watershed. Many of the benchmark watersheds also support several overstory species and are subject to a variety of influences not typical of experimental watersheds (see table A.1). The variety of influences limits the comparison between water quality measurements from the benchmark watersheds and the experimental watersheds. The pretreatment conditions described in the following chapters are based on measurements from experimental forests (the control). However, the 43 beachmark basins provide an initial idea of the range and variability in water quality from relatively undisturbed forest and rangeland watersheds across the United States.

Mean annual measurements for 8 constituents at the 43 gauging stations are listed in table A.2 in the appendix. Most of the measurements were taken from about 1965 to 1988. Figure 9 shows the distribution across the stations in mean annual level of each of the 8 constituents (the stations were ordered by constituent level for each graph). For all but temperature, 50% or more of the range in the level of the constituents is attributable to a few unusual stations. This fact is most noticeable for suspended solids, where the concentrations for all but 5 stations fall within the first 10% of the range in concentration. Tables A.1 and A.2 in the appendix show that most of the stations producing higher levels of suspended solids, dissolved solids, conductivity, pH, and bicarbonate drain rangeland rather than forested watersheds.

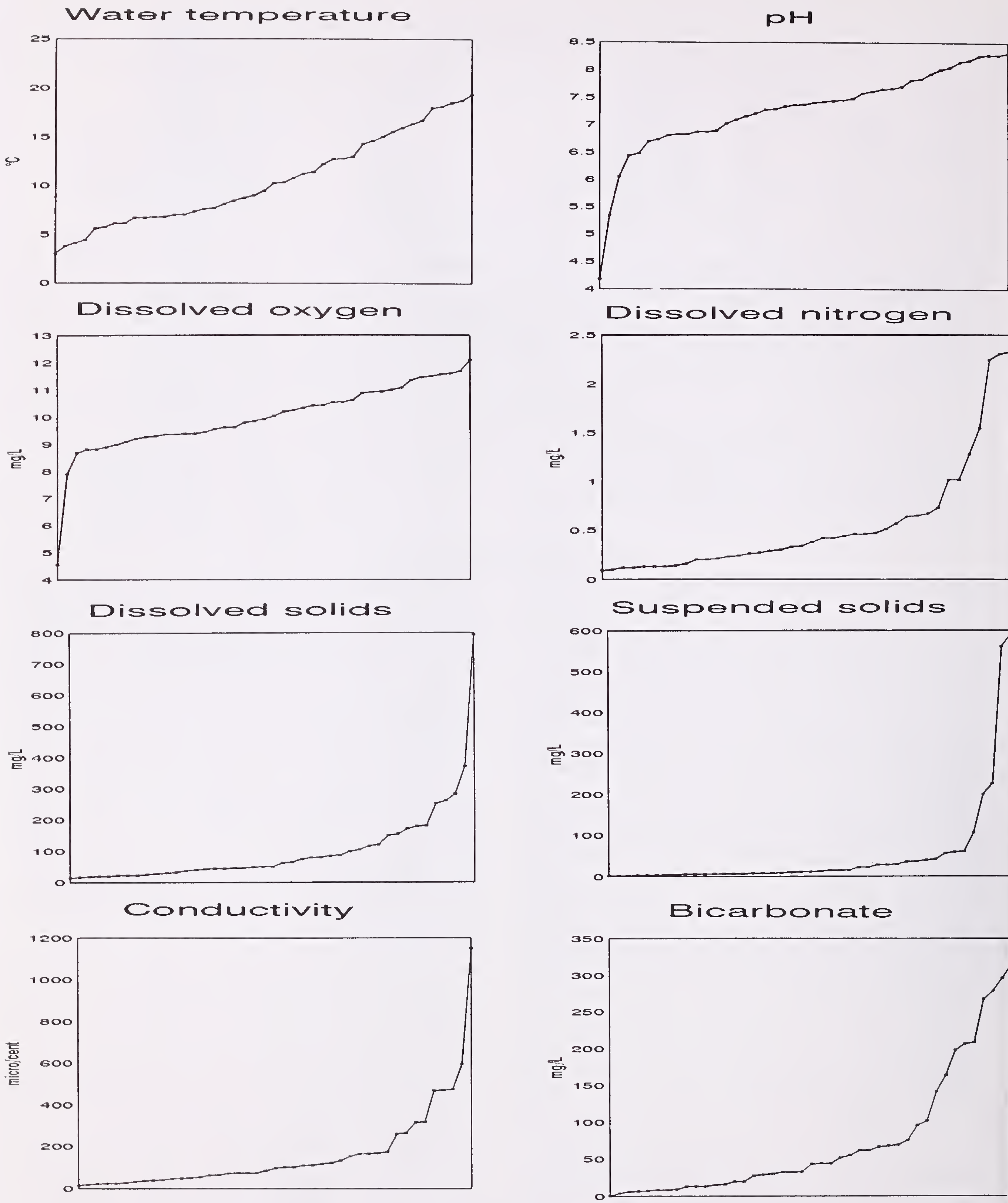


Figure 9. Mean annual levels of eight water quality constituents at 43 benchmark watersheds listed in table A.2. Watersheds arranged along horizontal axis in order of increasing concentration or level.

Chapter 3

Southeast

About 40% of the southeastern United States is forested, with the proportion ranging as high as 65% in the states bordering the Atlantic Ocean (USDA Forest Service 1982). Of the total forested area of about 88 million ha, 30% is classified as pine forests (longleaf, slash, loblolly pines, and shortleaf pines), with various hardwood mixtures (particularly oaks and hickories) comprising the balance. The average timber productivity is higher in the Southeast than in any other region, with over 75% of the forests showing increments of more than 3.5 m^3/ha annually; the best 15% of the forests achieve rates of more than 8.4 m^3/ha annually. Rangelands comprise 19% (42 million ha) of the Southeast, overlapping in part with forestlands. Forage production varies greatly across the region, ranging from 500 kg/ha annually in the drier portions to the west to more than 5000 kg/ha annually in the wet grasslands of Florida. Ownership of forests and rangelands is largely private, with the federal government owning about 6% of these areas.

The topography of the Southeast ranges from extensive coastal plains near the Atlantic Ocean and Gulf of Mexico (< 50 m elevation), up through rolling piedmont terrain (50-200 m elevation), into the southern Appalachian Mountains (up to 2000 m elevation). Precipitation exceeds 1500 mm/yr in the mountains and along the Gulf Coast, and it averages about 1200 mm/yr over most of the rest of the region. Runoff averages about 350 mm/yr across the region and up to 800 mm/yr in the wettest areas. Water quality varies substantially across these physiographic areas, as do responses to forest practices.

The impact of European settlement on land use has been dramatic. Much of the area was deforested and cropped intensively for varying periods before reversion back into forests. As an illustration of the magnitude of these land use impacts, the average amount of soil eroded from the entire piedmont was more than 15 cm (≅1500 Mg/ha) (Trimble 1974). Sediment is the most important pollutant in southern waters (Marion and Ursic 1992).

Coweeta Hydrologic Laboratory, North Carolina

The Coweeta Hydrologic Laboratory was established within the Nantahala National Forest in 1931 (Douglass and Hoover 1988) and is one of the most intensively studied sets of watersheds in North America. The first director, Dr. Charles R. Hursh, set the objective for Coweeta: to determine the impact of forest condition and agricultural land use on erosion, stream water supply, and stream water distribution. Labor for the construction of facilities at Coweeta was provided by various government work programs, and by 1936, Coweeta had 16 stream gauges operating. Watershed treatments have included burning and conversion to corn cropping, woodland grazing, and forest harvest (with and without log removal).

Geography and vegetation

Coweeta is in the eastern part of the southern Appalachian Blue Ridge chain. Most soils belong to either the Inceptisol order (relatively young, poorly developed horizons) or Ultisol order (older, highly weathered). Inceptisols (Dystrochrepts and Haplumbrepts) are generally found where mass movement has resulted in younger soils (Velbel 1988); Ultisols (Hapludults) occur where materials have remained in place for thousands of years. Many of the soils in the basin are underlain by highly weathered rock called saprolite. The original rocks at Coweeta had a density of about 2.8 kg/L, compared with only 1.6 kg/L for saprolite, indicating the great extent of weathering. The ridgetops have from 6 to 23 m of saprolite overtop unweathered bedrock, and the rate of saprolite formation is about 40 mm/1000 yr. The major minerals forming soils in Coweeta are quartz, biotite, and muscovite micas; plagioclase feldspar; and almandine garnet. The watersheds are thought to be underlain by relatively watertight bedrock.

The overstory vegetation in the Coweeta Basin historically consisted of an oak/chestnut forest, but the loss of chestnut to the exotic chestnut blight fungus has shifted the forests into an oak/hickory class (Day et al. 1988). Aboveground forest biomass is in the vicinity of 140 mg/ha for control watersheds, with aboveground net primary productivity of about 8 mg/ha. Both values place the Coweeta forests below average values reported for other hardwood forests in the South (Monk and Day 1988). The recovery of net primary productivity is very rapid following clearcutting. Aboveground net primary productivity reached about two-thirds of the preharvest level in the third growing season after clearcutting (Boring et al. 1988).

Climate

Average daily air temperatures (Swift 1988) span from about 4° C in January to 20° C in July, with average daily solar radiation ranging from about 8 MJ/m^2 per day in December to 19 MJ/m^2 per day in June. Solar radiation depends strongly on slope and aspect, and temperature varies strongly with elevation, slope, and aspect. Precipitation increases about 3-5% for every 100 m increase in elevation, ranging from a long-term average of 1800 mm/yr at 690 m to about 2400 mm/yr at 1360 m. Precipitation is well distributed through the year, with the driest month (October) averaging about 120 mm of rain at low elevation. Many storms last only a short period (45% of storms last less than 3 hours) and yield a low amount (half the storms drop less than 5 mm of precipitation). This combination leads to a relatively high proportion (about 20%) of annual precipitation lost to canopy interception. Snow comprises only 2-10% of precipitation at lower elevations. Evapotranspiration (including interception) removes about 900 mm/yr from low elevation watersheds and about 500 mm/yr from high elevation watersheds.

Hydrograph

Streamflow at Coweeta is closely coupled with precipitation. Low elevation watersheds (WS 2 and WS 18) release about half of precipitation as streamflow, with about 5% of rainfall showing up in streams as stormflow (fig. 10). High elevation watersheds (WS 36 and WS 27) with steep, shallow soils yield about 75% of rainfall to streamflow, with about 20% as stormflow. High elevation watersheds have greater runoff and greater frequency of high-runoff events (fig. 11). The annual hydrograph is dominated by high flows in spring (fig. 12), when precipitation is high and transpiration is low. Flows are reduced through high transpiration rates in summer and low precipitation rates in autumn.

Baseline water quality

The chemistry of precipitation is dominated by H^+ (47% of total cations) and sulfate (63% of all anions).

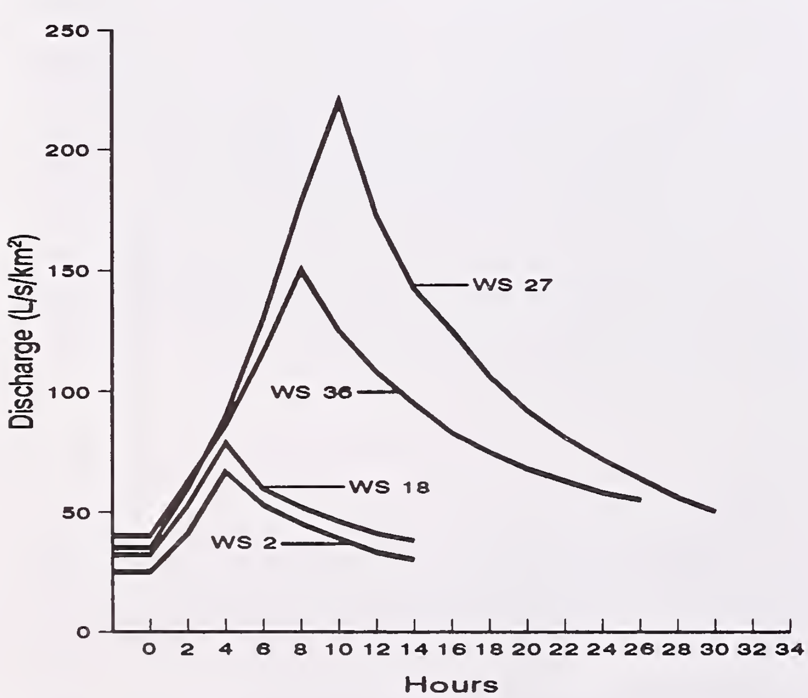


Figure 10. Storm hydrographs for control watersheds at Coweeta (from Swift et al. 1988).

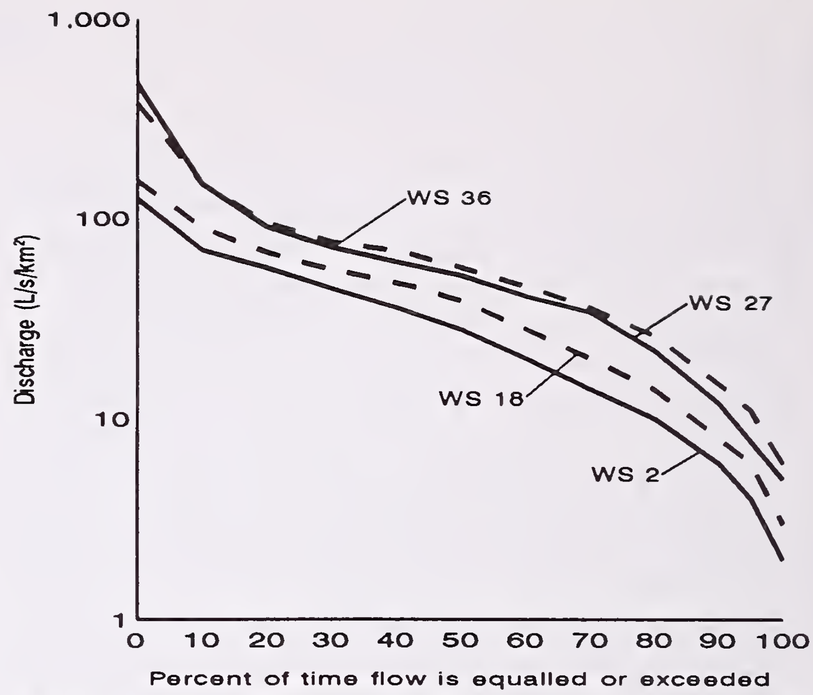


Figure 11. Flow frequency distribution for control watersheds at Coweeta (from Swift et al. 1988).

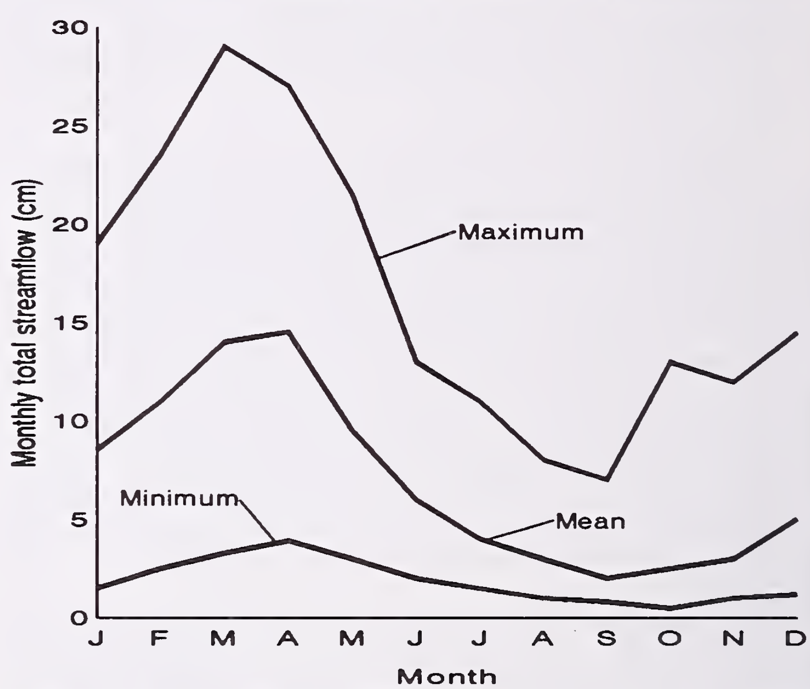


Figure 12. Mean monthly streamflow for control WS-2, compared with minimum and maximum observations (from Swift et al. 1988).

Rainfall is a dilute mixture of sulfuric and nitric acids, with an average pH of about 4.6 (Swank and Waide 1988). The chemistry of streamflow differs greatly from precipitation, with sodium, calcium, and magnesium comprising about 90% of the cations (in low elevation WS 2) and bicarbonate accounting for 75% of the anions. Stream water is characterized as a dilute bicarbonate salt solution with a pH of about 6.7. The total concentration of ions in WS 2 is about 110 µmolc/L (μmol_c/L = micromoles of charge per liter) each of cations and anions. At higher elevations, the total quantity of ions in stream water is about half that of low elevation streams (WS 27 has about 62 μmol_c/L each of cations and anions), and the concentration of sulfate matches that of bicarbonate.

Both nitrate and phosphate deposited in precipitation show net retention in the watersheds. Nitrate-N concentrations range from about 2.8 µg/L for WS 2 to 20 µg/L for high elevation WS 27. Phosphate-P concentrations are very low (3-6 µg/L). The export of suspended bedload averages about 260 kg/ha annually, and these materials account for more N loss (0.52 kg/ha per year) than dissolved losses (0.08 kg/ha per year).

Water quality responses to treatments

Twelve watersheds at Coweeta have received vegetation manipulation treatments, but water quality was not monitored in many of the earlier treatments. The impacts of many of the forest practices examined were too slight to warrant water quality concerns (Swank 1988). However, clearcutting hardwood vegetation increased export of nitrate-N (ranging from 15 to 700 µg/L) for periods of up to 20 yr. Clearcutting hardwood vegetation followed by planting to white pine has also increased nitrate-N exports for 25 yr. Responses to less intensive treatments, such as partial cutting of the forest, have had little effect on nitrate-N concentrations. A natural disturbance, defoliation by a fall cankerworm, led to large increases in stream water nitrate-N concentrations (exceeding 40 µg/L). Forest practices have had no substantial effect on the concentrations of other elements.

In the 1940's, the effects of cattle grazing were examined (Johnson 1952). Six cattle were grazed annually (from May through September) in a 59 ha watershed. After the first year, forage production was too low to sustain the cattle, and supplemental feeding was necessary. The cattle removed practically all hardwood trees less than 6 cm diameter in the lower portion of the watershed. Trampling by cattle increased the bulk density of the upper (0-10 cm depth) soil by 50% and decreased infiltration capacity by 90%. The diameter growth of dominant, upper canopy trees was decreased by 25-50%. After 9 yr of grazing, the hydrograph was much more responsive to storm events, and sediment concentrations were much higher (108 mg/L in one storm for the grazed watershed vs. 30 mg/L for the same storm in the control watershed). Recovery of hydrologic properties after grazing stopped was relatively rapid, requiring only a few years (Patric and Helvey 1986).

A great deal of effort also focused on the production of sediments from roads. Poorly constructed roads, such as those with road surfaces of bare soil and unvegetated cut and fill banks, can be major sources of sediment loads in streams (Swift 1988). During a 15-yr demonstration of "exploitative logging" impacts, unregulated harvest practices resulted in severe erosion of poorly designed roads (about 400 m^3/km of road length) and stream sedimentation (up to 5700 mg/L of sediment!). More conventional studies showed that bare road surfaces, for example, lost about 1.2 Mg/ha of surface to erosion for each cm of precipitation; a surface of grass or gravel lowered erosion to 0.05-0.65 Mg/ha. Unvegetated cut slopes lost about 150-360 Mg/ha of surface to erosion, compared with negligible losses from slopes seeded to grass. Researchers also developed a "broad-based dip" design for roads, where waterflow on the road surface is diverted downslope in a diffuse manner by tilting the slope of the road outward (by about 3%) for a 6 m stretch about every 60 m. This "best management practices" design has been adopted for forest road guidelines throughout the eastern United States.

The effects of a variety of forest treatments on stream water temperatures included converting forestland to mountain farmland, deadening riparian trees with herbicides, and clearcutting (Swift 1971). In the first year after conversion to farmland, summer stream temperature increased by a maximum of 12 °C; 8 yr later, the maximum difference was still 6 °C. First-year increases for the herbicide and clearcutting treatments were much lower, about 3 °C.

Great Smoky Mountains National Park, North Carolina and Tennessee

The climate and precipitation patterns of the Great Smoky Mountains National Park are similar to those of Coweeta. Silsbee and Larson (1982) characterized water quality in 28 major drainages within the Park. The concentrations of nitrate-N increased with elevation, from 0.5 mg/L at 500 m to 4.9 mg/L at 1500 m. The authors divided the sampling locations into two groups: those that had less than 25% of the area logged prior to Park establishment in the 1930's, and those that had more than 75% of the area logged. Nitrate concentrations were substantially lower (50-75%) in streams draining the more heavily logged areas. Along with lower concentrations of nitrate, these streams had higher pH and alkalinity. The authors attributed the greater nitrate retention in second growth forests to greater accumulation of N in rapidly aggrading biomass pools in the younger forests. Water chemistry (in terms of nitrate concentrations) demonstrated a long-term response to harvesting. Short-term increases in nitrate concentrations could be balanced by longer-term decreases.

High-elevation forests in the Smoky Mountains appear incapable of retaining the high rates of nitrogen deposition from the atmosphere. Johnson and Lindberg (1992) showed that nitrogen leaching losses exceeded the high rates of nitrogen deposition (about 25 kg/ha annually) in both spruce and beech forests. Nitrate-N concentrations in mineral soil solutions averaged about 1.3 mg/L for a red spruce stand and about 2.1 mg/L for a beech stand (with much higher peaks). Any forest practices (such as harvesting) may have dramatic effects on nitrate leaching from such heavily loaded forests; these effects could include higher nitrate concentration soon after harvesting, followed by reduced nitrate concentrations after forest recovery. The role of riparian ecosystems in moderating these changes could be very important.

Walker Branch Watershed Project, Tennessee

The Walker Branch Watershed Project was initiated in 1967 under sponsorship of the U.S. Atomic Energy Commission with 3 primary objectives (Van Hook 1989): to provide baseline values for unpolluted natural waters; to contribute to knowledge of cycling and loss of chemical elements in natural ecosystems; and to enable construction of models for predicting the effects of man's [sic] activities on the landscape. The watershed is located in the Ridge and Valley geographical province in east Tennessee on the U.S. Department of Energy's Oak Ridge Reservation.

Geography and vegetation

The watershed occupies a total of 97.5 ha in two subcatchments; the top of the watershed is at 350 m elevation, and the weir is at 265 m elevation. The bedrock consists of siliceous dolomite, covered by up to 30 m of saprolite and soil. Soils are primarily Typic Paleudults, low in nutrients and pH. The bedrock does not form a watertight seal to the watershed (Luxmoore and Huff 1989).

The overstory vegetation is mostly oak-hickory type, with scattered Virginia and shortleaf pines on the ridges and tulip poplar and beech in the riparian zones (Johnson 1989). Aboveground tree biomass ranges from about 140 to 200 Mg/ha across the forest types in the watershed, and aboveground net primary productivity (estimated by summing woody increment, litterfall, and mortality) is about 9 Mg/ha per year (Edwards et al. 1989).

Climate

The average annual temperature for Walker Branch is about 14.5 °C, with an average of 4.4 °C in January and 25.1 °C in July. Precipitation averages about 1400 mm/yr, with a peak in winter (about 150-250 mm/month), a somewhat dry spring and early summer (50-100 mm/month), and moderate rainfall in mid- to late summer (100-150 mm/month) (Luxmoore and Huff 1989). Some precipitation occurs on about one-third of all days, with rates of < 10 mm/day for 55% of the storms and < 30 mm/day for 90% of the storms. Maximum recorded daily precipitation was 140 mm. Interception losses (145 mm/yr) account for about 20% of evapotranspiration (745 mm/yr), or about 10% of total precipitation.

Hydrograph

The watershed is comprised of a West Fork and an East Fork; both forks are intermittent in flow in the upper reaches. The perennial reaches are sustained by prominent springs. Annual streamflow for the entire watershed averages about 720 mm/yr. The combined streamflow plus estimated evapotranspiration exceed precipitation by about 10%, indicating the true area of the watershed is larger than the surface topographic area of the watershed because of subsurface flow into the watershed.

Baseline water quality

A great deal of effort has been devoted to developing methods for measuring atmospheric deposition at Walker Branch (Lindberg et al. 1986). Precipitation is a dilute mixture of strong acids (pH 4.0-4.2), dominated by H^+ and sulfate and nitrate anions. Wet deposition contributes only one-third to one-half of the total deposition of ions. Within-canopy buffering of the acids results in replacement of most of the H^+ in throughfall with potassium and calcium. The canopy retains about half the nitrate.

In contrast to the acidic precipitation, stream water is a well-buffered bicarbonate salt solution. Calcium and magnesium are the dominant cations, and bicarbonate is the dominant anion. Total concentration for cations and anions is about 1500 μmol_c/L each, more than 10 times the concentrations found at Coweeta. The pH of stream water remains relatively high, despite substantial deposition of acidity and decrease in soil base saturation over time (Johnson and Henderson 1989). Nitrate-N concentrations are moderate, about 0.8 mg/L.

Water quality responses to treatments

No treatments have been applied to the Walker Branch Watershed, but a harvesting study was conducted in a set of 5 mini-watersheds (0.25-0.54 ha) nearby (Johnson and Todd 1987). One mini-watershed was retained as a control, and 4 were harvested. Only sawlogs (>28 cm diameter) were removed from two mini-watersheds, and all above-stump materials were removed from the other two. Lack of streamflow in the mini-watersheds prevented examination of the effects of harvesting on water quality, but soil solution concentrations were assessed. The tree harvesting treatment increased the nitrate-N concentrations in soil solution from near 0 to a maximum of 0.14 mg/L, well below any threshold of concern about water quality.

Santee Experimental Forest, South Carolina

Climate of the Santee Experimental Forest is humid and subtropical with hot summers and short, mild winters (Richter et al. 1983). Precipitation averages about 1350 mm/yr, with more than 60 mm falling in the driest months (April and November). The chemistry of precipitation is dominated by base cations (50 $\mu mol_c/L$) and chloride (30 $\mu mol_c/L$). Richter et al. (1982) reported the results of prescribed fire treatments on water quality from a 160 ha watershed on the lower Coastal Plain in the Francis Marion National Forest in South Carolina. Topographic relief is less than 6 m across the watershed, and the soils are primarily acidic, infertile Aquults. Stream chemistry is relatively dilute, with only 14 µg/L of nitrate. At the time of the burning experiments, the forest comprised old, natural stands of loblolly pine (which were later destroyed by Hurricane Hugo in 1989). The watershed was divided into 20 management compartments of about 7 ha each, leaving 20-m buffer strips between the compartments and the streams. Over a period of 3 yr, 12 of the 20 compartments were burned in summer or winter. The fires consumed about 2.5 to 7.5 Mg/ha of forest floor materials, removing about 10 to 40 kg-N/ha. A series of 32 groundwater wells were sampled through the period, and no effects of fire were detected. Stream water sampling showed no signs of any impacts of the fire on water quality; the specific conductances (total ionic concentrations) were modestly greater in a control watershed than in the partially burned watershed (both in the range of 4 to 10 milli Siemens per meter). These authors concluded that careful use of prescribed fire could achieve vegetation manipulation objectives (such as reduction in understory hardwoods) with no impact on water quality. Mechanical and chemical alternatives might have substantially greater impacts than fire on water quality.

Georgetown County, South Carolina

Askew and Williams (1986) examined the impacts of conversion of a 2400 ha watershed from natural hardwood forest to loblolly pine plantations. Fourteen storm periods were sampled over a 2-yr period, with water collected from a variety of subwatersheds in different stages of the conversion operation (from hardwood control, to ditching to increase drainage, to old pine stands). Nitrate-N concentrations remained low in all stages, reaching a maximum of about 1 mg/L when drainage ditches were installed. Dissolved oxygen increased marginally from the hardwood control forest (5.1 mg/L) to the older pine forest (6.9 mg/L).

Carteret County, North Carolina

Weyerhaeuser Company is conducting two watershed studies near the coast of North Carolina (Hughes et al. 1989). The first study is examining the quantity, timing, and quality of water draining from forest plantations into the Isaac Creek Estuary. The forest plantations were established on pocosin wetlands (raised bogs with poor drainage), using intensive site preparation techniques (including ditching and riser dams to control waterflow). Annual precipitation averages about 1380 mm/yr, and runoff averages about 275 mm/yr (20% of precipitation). Runoff from the plantation averaged about 13 mg/L of suspended sediment (about half of this is dissolved and particulate organic matter). Turbidity averaged about 20 NTU. The concentration of inorganic N (ammonium + nitrate) averaged about 0.1 mg-N/L. The only notable effect of plantation establishment on the estuary was some accumulation of sediment at the head of the estuary following ditch construction. No impacts on fish or shrimp resources was noted.

The second Weyerhaeuser study is focusing on manipulating water tables and controlling waterflow from ditch outlets that drain the plantation. The experiment aims to examine: open ditch drainage (without riser dam control); optimal water supply for trees (draining in winter, retaining runoff in summer); and drainage aimed at optimal seasonal flow of freshwater into estuaries. At present, only information on water quality draining the unmanipulated plantation is available. Turbidity ranges from about 1 to 20 NTU, and nitrate-N concentrations vary from near 0 to 1 mg/L. Total coliform counts are generally low, < 10 colonies/100 mL, but occassionally up to 60 colonies/100 mL. For comparison, the total coliform count for water flowing in a highway ditch (emptying into the same drainage system) had a count of over 2000 colonies/100 mL because of sewage from homes along the road. Baseline water quality from forest plantations in coastal North Carolina appears very good.

Bradford County, Florida

Riekirk (1983) contrasted the effects of low-intensity forest harvesting and planting on water quality with those resulting from harvesting coupled with intensive site preparation. The "flatwood" sites were nearly level, with slopes < 0.1% at an elevation of about 44 m above sea level. The soils (Plinthic Paleudults) consist of sand (0.5 - 2.0 m deep) overlying a thick clay horizon (1-4 m deep). The fluctuating water table is usually within 0.5

m of the soil surface. The climate is mild and wet, with about 1400 mm of precipitation. The vegetation is primarily slash and longleaf pines, with some bald cypress in ponds within the watersheds. Precipitation during the 3-yr study ranged from about 1250 mm/yr to 1540 mm/yr, with chemistry dominated by base cations (20 μmol_c/L) and probably chloride (data not given).

The experimental watersheds were isolated by building ditches and elevated roads around the perimeters of 3 small catchments (48-137 ha). The pine forests were harvested from two of the watersheds, leaving the cypress/hardwood stands undisturbed. The minimum impact treatment involved bole-wood harvesting, chopping the residual slash and understory, bedding the soil, and planting pine seedlings. The maximum impact treatment also included burning, windrowing slash, and harrowing the soil.

Both harvesting treatments had relatively minor influences on water chemistry at the sampling flumes. Stream water pH rose slightly in the maximum treatment, from about 3.8 in the control watershed to 4.2. Nitrate-N concentrations increased from control values of less than 40 µg/L to 1000 µg/L. Sediment loads were low in all treatments (after the boundary roads were graded to drain away from the watersheds), averaging about 14 mg/L. The impacts of these treatments on water quality were too slight to warrant concern relative to water quality standards.

Palustris Experimental Forest, Louisiana

The effects of grazing and burning, including their effect on soil physical properties, were examined in a 12-yr study by Linnartz et al. (1966). Moderate grazing significantly increased soil bulk density by about 5% down to a depth of 15 cm, and by 2% in the 15-25 cm depth; the increases for heavy grazing were 7% and 4% for these depths. Total pore space was reduced by 5-8%. Maximum infiltration rates for ungrazed, moderately grazed, and heavily grazed plots were 57 mm/hr, 35 mm/hr, and 26 mm/hr for 30 minutes. Storms with intensities of 40 mm/hr for 30 minutes (or greater) occur annually on average, indicating substantial opportunity for overland flow on grazed areas. No signs of widespread erosion on grazed areas were apparent, however.

Grant Memorial Forest, Georgia

This study was established to examine the effects of commercial clearcutting and planting operations on water quality in the Georgia Piedmont (Hewlett et al. 1984). Two watersheds (32.5 ha treatment, 42.5 ha control) were gauged and calibrated for 1 yr in 1974. The watersheds had been heavily farmed and eroded before 1950, when abandonment led to a succession of pine and hardwood species. At the time of the study, the dominant vegetation was loblolly and shortleaf pines, with patches of oaks, hickories, sweetgum, and tulip poplar. The soils are predominantly Typic Hapludults (highly weathered and very acidic), and rainfall averages about 1310 mm/yr.

The smaller watershed was logged in 1975. Rubber-tired skidders hauled logs to landings. A 10-15 m wide buffer strip was retained along streams. The locations of roads, skid trails, and landings were left to the logger's discretion, resulting in a haphazard pattern typical of current practices in the Piedmont. The site was then roller-chopped (a large, rotating drum with blades pulled behind a Caterpillar tractor to crush remaining hardwood stems and slash) twice in 1975. In 1976, the site was planted by machine with loblolly pine. The planting tractor included a 2-m wide V blade in front that cut about 0.15 m deep (to the B horizon) over about 50% of the watershed area.

Harvesting produced no notable increase in stream water nitrate-N concentrations. Concentrations averaged less than 15 µg/L in both the harvested and control watersheds. Sediment concentrations and turbidity were not assessed. The most dramatic effect of harvesting was the extreme increase in summer maximum temperatures (11 °C) (Hewlett and Fortson 1982) and decrease in winter minimum temperatures (6 °C lower than the control). A 12 m buffer strip had been retained along the stream, which was expected to provide sufficient cover to protect stream temperatures. The investigators concluded that greater than expected differences in temperatures might reflect less buffering of stream temperature by the low inputs of groundwater (cool in summer, warm in winter) along the reach of the stream flowing through the clearcut. Any impacts on fish were not known.

Brushy Ridge Branch, North Carolina

Stream temperatures were also examined in a harvesting experiment at Brushy Ridge Branch, near Brevard, North Carolina (Swift and Baker 1973). Stream temperatures were about 2 °C higher in a fully exposed, 600 m stretch of the stream than in the uncut forest. Temperatures declined rapidly below the exposed stretch, probably as a result of inputs of cool groundwater.

Ouachita Mountains, Arkansas

Miller et al. (1988) examined the effects of logging on water quality in the Ouachita Mountains, about 35 km north of Hot Springs, Arkansas. The watersheds ranged in size from 4.2 to 5.9 ha, with slopes of 15 to

30%, and contained intermittent streams. The soil parent materials are sandstones and shales that are extensively folded and faulted. Soils are Typic Hapludults. Nine watersheds were examined, with three clearcut, three selectively logged, and three uncut. The vegetation on the watersheds was not described. Logs were skidded uphill to landings, with no restrictions on operations near stream channels. In the clearcut, residual vegetation was drum-chopped and burned in winter; no site preparation treatment was applied in the selection cut.

Clearcutting increased sediment yields by 20-fold in the first year, declining to 6- and 2.6-fold increases in the following 2 yr. Selective cutting increased sediment yields by about 2-fold the first year, and less than 2-fold in subsequent years. Despite the large relative response in sediment yields, the maximum observed annual rate of 0.2 Mg/ha for the clearcut watershed (first year after cutting) was relatively small.

Natchez Trace State Forest, Tennessee

Eight mini-watersheds, ranging in size from 0.17 to 0.56 ha, were used to examine the effects of clearcutting on water quality near Lexington, Tennessee (McClurkin et al. 1985). Four were harvested and four retained as controls. Vegetation was predominantly loblolly pine, ranging in age from 29 to 37 yr old. Soils are Typic Paleudalfs (clayey B horizon, moderately acidic) and Typic Fragiudalfs (with a hard fragipan layer). Clearcutting used rubber-tired skidders to haul logs along the contour to the nearest ridge and then to a landing. Haul roads and landings were located outside the mini-watersheds, and the forest floor remained generally intact across the sites. After harvest, the sites were replanted with loblolly pine seedlings with no site preparation.

Harvesting increased the stream sediment concentrations for stormflow peaks from 82 mg/L for the control watersheds to 183 mg/L for the harvested watersheds. After about 3 yr, sediment concentrations returned to values expected for undisturbed watersheds in the region. Clearcutting produced no changes in dissolved Kjeldahl nitrogen (ammonium + organic nitrogen; the digestion procedure was not modified to include nitrate [Duffy 1985]). Annual average concentrations remained below 0.14 µg/L for control and harvested watersheds. Suspended sediments contained about 6 mg N/kg of sediments, and sediment transport accounted for about one-third of the N losses on control watersheds and two-thirds on harvested watersheds. These hydrologic outputs of N were less than precipitation inputs.

Tallahatchie Experimental Forest, Oxford, Mississippi

A massive tree-planting program was developed in the late 1940's in parts of northern Mississippi to revegetate and stabilize lands that had been severely eroded. By the early 1980's, about 340,000 ha of loblolly pine forests had been established in a 19-county area (McClurkin et al. 1987). Many of the older plantations have reached maturity, and McClurkin et al. (1987) developed a research project to examine the effects of thinning and harvesting on water quality. The soils of their study site were developed in silty loess (wind-blown) deposits on the ridges and upper slopes. Lower slope sites had loamier soils typical of the Coastal Plain. A variety of soil series in the plots belonged to Ultisol and Alfisol orders; slopes ranged from 12 to 20%. Replicated, 0.8 ha plots were established in a 21-year-old pine stand, and 3 treatments were applied: control, thinning to a basal area of 16 m^2/ha, and clearcutting and replanting with loblolly pine seedlings. Harvested material was cut into small lengths and hand carried from the plots. The design explicitly avoided confounding harvest effects with those from skidding, loading, and hauling of logs from the site. One runoff plot (1.8 m by 11 m) was installed in the center of each treatment plot, parallel to the slope. These were constructed with thin-gauge sheet metal driven into the ground to form the borders, with a metal trough at the lower end to collect surface runoff. One tension-free lysimeter was installed near each runoff plot at a depth of 0.15 m in the mineral soil.

After 2 growing seasons, reduced litterfall and accelerated decomposition lowered the biomass of the forest floor in the thinned plots (21.3 Mg/ha) and clearcut plots (15.2 Mg/ha) relative to control plots (23 Mg/ha). The effect of harvest on surface runoff differed among blocks; one block with high clay content at the surface showed increased runoff for the clearcut plot, while the other blocks showed no treatment effects. Sediment concentrations in runoff were not affected by treatments, averaging 102 mg/L for all treatments and years. Total Kjeldahl nitrogen (ammonium+organic N) concentrations were relatively high for surface runoff in the control plots (1.1 to 2.1 mg/L) and increased by about twofold after clearcutting. Nitrogen concentrations in lysimeter water at 0.15 m depth were lower, averaging about 0.5 mg/L for control plots and 0.6 to 1.2 mg/L for the clearcut plots. McClurkin et al. (1987) noted that the concentrations of nitrogen in runoff was much higher than that found at a watershed scale on similar soils. They reported that little of the plot runoff would be expected to reach streams. They concluded that clearcutting without disturbance of the forest floor does not significantly increase sediment losses or nutrient concentrations in soil solution.

Upper Coastal Plain, Mississippi

Beasley (1979) examined the effects of intensive site preparation on sediment losses from 4 relatively steep watersheds. The soils belonged to several series of Ultisol and Alfisol, with slopes averaging 30% and ranging up to 50%. Before harvest, the watersheds supported mixed forests of hardwoods and shortleaf pine; most trees were pulpwood-size. Harvested logs were winched uphill to ridges and then pulled to landings by rubber-tired skidders. Four levels of site preparation were applied (one in each watershed):

1) Control—fertilized with phosphorus and potassium, limed, seeded with subterranean clover seed; and planted with loblolly pine seedlings.
2) Chop and burn—control treatments, plus chopping of residual vegetation, piling slash with bulldozers and burning.
3) Shear and windrow—control treatments plus shearing of standing vegetation at ground level with a V-blade, and piling of slash and topsoil into windrows *into the stream channels* to be burned.
4) Bedded—shearing and windrowing treatment, plus plowing to produce raised beds (2.4 m!) along the steep slope contours at intervals of 3 m (almost a terracing treatment).

Immediately after site preparation, bare soil was exposed on 37% of the chop and burn watershed, 53% of the shear and windrow watershed, and 69% of the bedded watershed (data not given for control). Clover decreased soil exposure in the first year. By year 3 exposed soil was reduced to 4%, 10%, and 16% on the chop and burn, shear and windrow, and bedded watersheds. Treatments greatly increased stormflow from the watersheds, from an average of about 28 mm/yr for the control watershed (harvested, but less intensively treated) to more than 400 mm in all intensively prepared watersheds in the first year. Sediment concentrations for the first year after treatment ranged from 525 to 2100 mg/L in stormflow from the control watershed and up to 15,000 mg/L in the bedded watershed. Total sediment losses for the first 2 yr after treatment were 0.7 Mg/ha for the control, 15 Mg/ha for both the chop and burn and shear and windrow, and 20 Mg/ha for the bedded. Beasley (1979) concluded that all 3 intensive site preparation treatments increased sediment runoff similarly and that bedding on such steep slopes was not very practical.

Ursic (1970) examined how streamflow patterns and sediment concentrations are affected by a combination of light burning of the forest floor, herbicide application to the surviving trees, and underplanting with loblolly pine seedlings. Total streamflow from the treated watersheds was increased by about 25% over the control, and sediment yield averaged 22-54% greater than from control watersheds. Sediment yield increased by 40-120%, but even the highest increase resulted in an annual loss of only 0.5 Mg/ha. Ursic (1970) concluded the increase in sediment loss was not alarming, and that rapid recovery of vegetation practically eliminates soil movement after several years.

Ursic (1991) also examined the effects of harvesting a mixed hardwood forest (primarily white and red oaks) using rubber-tired skidders and a cable-yarding system. Three catchments (1.4-1.9 ha) in the Holly Springs National Forest in north Mississippi were monitored beginning in 1959. Slopes average 8-12%, with maximum slopes of 30-60%. Annual precipitation averaged 1390 mm/yr, with runoff averaging between 280 and 390 mm/yr for the 3 watersheds. Soils near the ridges were Typic Fragiudults, and those closer to the streams were Typic Fragiudalfs; midslopes contained Thermic Paleudults and Vertic Hapludalfs. One watershed was harvested (in fall 1982), trees limbed, and boles yarded along the contour to skid roads using rubber-tired tractors; skidders were not allowed to cross the stream. The other watershed was harvested similarly, but boles were yarded with a cable system mounted on a tractor outside the watershed boundary. Both watersheds were planted with loblolly pine in the spring of 1983, and residual hardwoods (>25 mm dbh) were injected with herbicides. Pine survival was low, and the watersheds were replanted in 1984.

Sediment concentration in the Holly Springs study averaged about 70 to 90 mg/L before harvesting, representing annual yields of 0.2 to 0.3 Mg/ha. The first year after harvest, sediment concentrations were high for all 3 watersheds: 230 mg/L control, 130 mg/L for tractor-skidded, and 640 mg/L for cable-yarded. In the second post-harvest year, sediment concentrations were low for the control (30 mg/L) but high for the harvested watersheds (230 mg/L tractor-skidded, 1630 mg/L cable-yarded). In later years, sediment concentrations were similar among watersheds except for high values in the cable-yarded watershed in the fifth year after harvest.

Summary

These small-watershed studies demonstrated that harvesting generally has no substantial effects on stream water chemistry but that intensive site preparation has the potential to greatly increase sediment loads, especially in steep terrain (table A.3, in the appendix). The major focus of concern is sedimentation associated with roads and intensive site preparation treatments (Goetzl and Siegel 1980; Yoho 1980).

Chapter 4

Northeast

About 60% of the northeastern United States is forestland (USDA Forest Service 1982) comprised of a wide range of forest types. Hardwood forests dominate the landscape, with mixed forests of maple, beech, and birch comprising about 30% of the forests. Oak/hickory forests comprise another 20%. Conifer forests comprise about 25%, ranging from loblolly and shortleaf pines at low elevations, to white, red, and jack pines at middle elevations, to spruce/fir forests at high elevations. The average timber productivity for the region is between 3.5 and 6.0 m^3/ha annually; about 8% of the forests surpass 8.4 m^3/ha annually. Most forestland is privately owned, with federal ownership comprising less than 5%. Rangelands are uncommon in the Northeast, covering only about 0.1% of the region (mostly in New Jersey and Maryland).

The topography of the Northeast spans from coastal plains near the Atlantic Ocean (<50 m elevation), through piedmont plateaus (50-200 m), into the folded mountains of the Appalachians (200-1800 m). Precipitation averages about 1000-1500 mm/yr across most of the region, with runoff averaging about 300 mm/yr at lower elevations and 1000 mm/yr in the mountains.

Most of the forests of the region have been harvested at least once since European settlement, frequently with intervals of agricultural management (Williams 1989).

Hubbard Brook Experimental Forest, New Hampshire

The Hubbard Brook Experimental Forest was established in 1955 within the White Mountain National Forest as the principal research area for the management of New England watersheds (Likens et al. 1977; Bormann and Likens 1979). A group of researchers from Dartmouth College (G. Likens, H. Bormann, and N. Johnson) began examination of watershed-level nutrient budgets in 1963 in cooperation with scientists from the USDA Forest Service (particularly R. Pierce). The objective of the early studies was to determine the magnitude of the biogeochemical flux and internal cycling of nutrients in northern hardwood ecosystems in the White Mountains of New Hampshire. Watershed sizes range from 12 to 76 ha.

Geography and vegetation

The landscape at Hubbard Brook emerged from the melting glacial ice between 12,000 and 13,000 yr ago. The bedrock underlying the experimental watersheds is highly metamorphosed sedimentary rocks (mudstones and sandstones) of the Littleton formation. Across most of the watershed areas, the bedrock is covered by a blanket of glacial till derived from the Littleton formation. The bedrock is thought to be watertight, with almost all water leaving the watershed via the streams. The soils are well-drained Spodosols (accumulations of iron, aluminum, and humus in the B horizon), with variable depths that average about 0.5 m. Soil pH averages about 4.5 or less.

The principal species comprising the northern hardwoods vegetation type are beech, sugar maple, yellow birch, and lesser quantities of white ash and basswood. Young forests contain a high density of pin cherry. In the 60- to 70-yr-old forests typical of the experimental watersheds, aboveground biomass averages about 250 Mg/ha, with annual aboveground net primary productivity of 10 Mg/ha. Following clearcutting, aboveground net primary productivity reaches preharvest rates in about 4-5 yr.

Climate

The continental climate conditions include average January and July temperatures of -9 °C and 19 °C. Precipitation averages 1300 mm (half of the years fall within 100 mm of this average) and is well distributed through the year. A snowpack of 1.5 m is common, but warm periods in winter occasionally melt the entire snowpack. Evapotranspiration (including interception) removes 500 mm/yr.

Hydrograph

Streamflow reaches a maximum with snowmelt in April, when 30% of annual runoff occurs (fig. 13). Streamflow is low in summer, with most water leaving the watershed via transpiration.

Baseline water quality

Precipitation chemistry has been followed continuously at Hubbard Brook for over 25 yr. Precipitation is an acidic solution, with H^+ accounting for about 70% of the total cation charge (Likens et al. 1977); sulfate accounts for about 60% of the anion charge. Over time, precipitation acidity has declined as sulfur emissions declined. Annual averages of precipitation pH ranged from 4.1 to 4.2 for 1965-1975, and 4.2 to 4.4 for 1980-1987 (Driscoll et al. 1989). The cation chemistry of stream water differs markedly from precipitation. Concentrations of H^+ in stream water are only about 15% of those in precipitation; inclusion of the concentrations of acidic Al^{3+} in stream water raises the concentration of acid cations in stream water to about 55% of that found in precipitation. The anion chemistry of stream water is

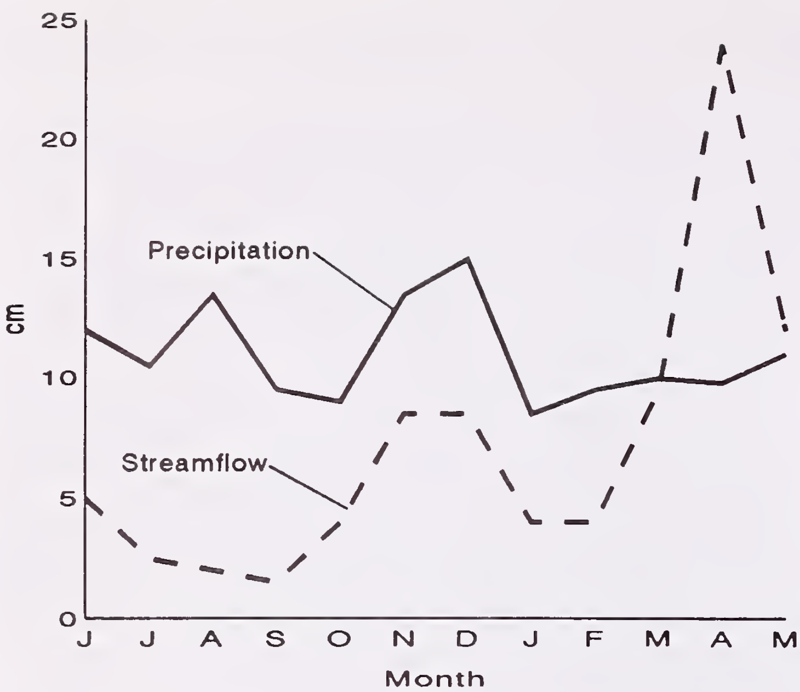


Figure 13. Average precipitation and streamflow for Watershed 6 at Hubbard Brook Experiment Forest for water years 1963-1974 (from Likens et al. 1977).

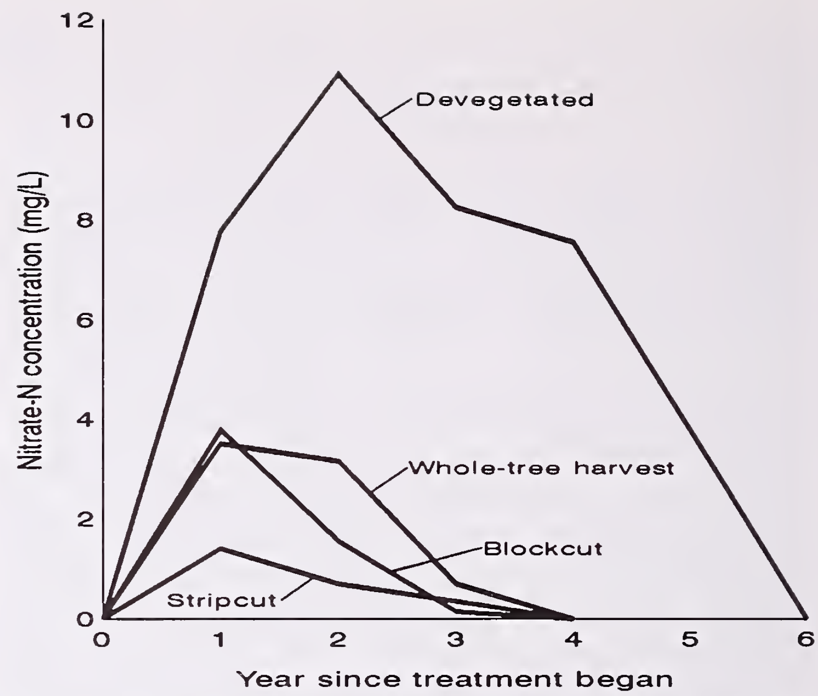


Figure 14. Nitrate-N concentrations for forest harvest treatments at Hubbard Brook (from sources cited in text).

similar to (though more concentrated than) precipitation, with sulfate and nitrate concentrations greatly exceeding those of other anions.

Phosphate deposited in precipitation is strongly retained within the forests; stream water outputs (stream water concentrations average <1 μg-P/L) represent about 20% of inputs. Nitrate is also conserved within the aggrading forests, but the balance is much closer with about 85% of the nitrate in precipitation matched by nitrate in stream water. Nitrate-N concentrations in stream water from undisturbed forests are moderately high, averaging about 0.5 mg - N/L. Losses of N as dissolved ions in stream water account for about 97% of total N losses (particulates = 3 %), whereas losses of P occur mostly (63%) in particulate forms (Likens et al. 1977).

Water quality responses to treatments

Three major watershed-level harvesting treatments have been investigated at Hubbard Brook: devegetation with no log removal (1965), conventional clearcutting in strips or an entire watershed (1970), and clearcutting with whole tree biomass removal (1983).

In the devegetation treatment (which does not resemble normal forest practices), all trees in Watershed 2 were cut in early winter, and revegetation was inhibited with herbicide applications for 2 yr. The treatment increased streamflow by 25-40% and had no effect on stream turbidity (and therefore none likely on sediment concentrations). Concentrations of stream water nitrate-N increased over 50-fold (fig. 14) (Likens et al. 1970), averaging about 15 mg/L. Stream water pH dropped by about 0.3 units, and aluminum concentrations increased by almost 10-fold. As revegetation occurred after herbicide treatments ended, nitrate concentrations returned to values similar to the control watershed by 1970 (5 yr after deforestation). The biomass of vegetation after 5 yr was only about half that expected for clearcut forests without herbicide treatments (Bormann and Likens 1979).

The strip-cutting treatment was applied over a period of 3 yr to Watershed 4; each year, one-third of the watershed was harvested in 25-m-wide strips perpendicular to the slope (Hornbeck et al. 1986, 1987). A buffer strip was retained along the stream to minimize direct impacts. The clearcutting treatment in Watershed 101 removed all trees from the entire watershed in one operation, with no buffer strip retained along the stream. In both cases, efforts were made to perform operations in an ecologically sound manner. All trees >50 mm diameter were cut, and merchantable boles were removed by rubber-tired skidders. The strip-cut treatment removed about 50 Mg/ha of biomass and left 83 Mg/ha of slash on the site. The clearcutting treatment removed 65 Mg/ha of biomass and left 85 Mg/ha of slash. Sediment accumulation behind the weir was not increased after logging, and turbidity was increased only in a few storm events (Hornbeck et al. 1987). Nitrate losses peaked in the first year after treatment began, with the completely harvested watershed showing more than double the nitrate-N concentrations (3.5 mg/L) of the strip-cut watershed. By year 3, the clearcut watershed had lower nitrate concentrations than the strip-cut watershed, and values for both watersheds returned to control values (or lower) by year 4. The nitrogen losses in stream water after harvest accounted for about 25-33% of the combined harvest losses in biomass removal and stream water.

Whole tree harvesting of Watershed 5 raised nitrate-N concentrations from about 0.05 mg/L before harvesting to 3.6 mg/L in the first year, declining back to (or below) control levels in the fourth year after harvest (G. Likens, Institute of Ecosystem Studies, Millbrook, NY, personal communication; see fig. 14).

Nashwaak River, New Brunswick

This study in central New Brunswick examined the effects of clearcutting on water quality (Krause 1982). The harvested watershed was 390 ha, ranging from 225 to 418 m in elevation. The bedrock is argillite, with a layer of glacial till (0.5 to several m depth) derived from argillite, granite, diorite, and gabbro formations to the northwest. The soils range from Typic Haplorthods on hilltops and slopes, to Aquic Haplorthods on poorly drained areas and Typic Haplaquepts along the streams. Hardwood forests dominate hilltops and slopes, mixed hardwood and conifer forests dominate gentle slopes, and conifers dominate level terrain and along the streams. Annual precipitation is about 1355 mm, with about one-third as snow. Preharvest merchantable timber volumes were about 190 m^3/ha, evenly split between hardwood and conifer species. All merchantable trees (> 0.1 m diameter at breast height) were cut and removed; smaller stems were left intact or damaged to varying degrees by the harvesting operation. Nitrate-N concentrations in uncut watersheds averaged about 0.12 mg/L. After harvest, nitrate-N concentrations increased but remained low. The average was about 0.6 mg/L and maximum concentrations reached 1.3 mg/L.

Petawawa Research Forest, Ontario

Hendrikson et al. (1989) examined the effects of conventional harvesting and whole-tree harvesting on soil solution chemistry in a mixed hardwood (aspen and poplar) and conifer (red pine and white) forest. The whole-tree harvesting prescription involved removal of all woody biomass from trees > 1.3 m in height after leaf fall had occurred. The conventional harvest prescription had all stems > 9 cm in diameter removed. Regeneration on both units consisted primarily of aspen sprouts. Nitrate-N concentrations at 1 m depth in the soil averaged about 0.08 mg/L for the uncut watershed in the second year after harvest, compared with 0.6 mg/L in the conventionally harvested unit (concentrations declined to 0.13 mg/L in the third year). Whole-tree harvesting did not affect nitrate concentrations.

Integrated Forest Study Sites in the Northeast

The Turkey Lakes research cite in Ontario was included as part of the Integrated Forest Study on the effects of acidic deposition (Johnson and Lindberg 1992). No forest practices have been examined in this northern hardwoods forest, but the high concentrations of nitrate-N in soil solutions (2.9 mg/L at 65 cm depth) are noteworthy. These soil solution concentrations from undisturbed forests match those found for stream water in harvested sites at Hubbard Brook in New Hampshire. Harvesting these forests might lead to very high nitrate concentrations in stream water.

At the northern hardwoods forest in the Huntington Forest in New York, soil solution concentrations are far lower than at Turkey Lakes, averaging just 0.2 mg/L at 60 cm depth.

Nitrate-N concentrations in soil solutions are relatively low in the balsam fir/red spruce forest on Whiteface Mountain in the Adirondack Mountains (Johnson and Lindberg 1992), averaging 0.4 mg/L at 60 cm depth. The nitrate concentrations are far lower than those found in the same study for red spruce stands in the Smoky Mountains (Chapter 3).

Various Locations in New England

Martin et al. (1984) summarized the effects of conventional clearcutting practices on water quality in 38 watersheds from around New England. Vegetation types were described as conifer (white spruce, red spruce, black spruce, and balsam fir); northern hardwood; and central hardwood (oaks, hickories, and red maple). Stream water N concentrations for unharvested watersheds were near 0 for central hardwood forests, between 0.15 and 0.5 mg/L for the conifer forests, and 0.15 to 1.0 mg/L for northern hardwoods. Clearcutting from 20% to 100% of the watershed resulted in no nitrate increases in stream water draining central hardwood or conifer forests. Stream water nitrate-N concentrations for northern hardwood forests showed no effect of clearcutting if <70% of the watershed was harvested, and either no change or an increase up to 2.0 mg/L in completely harvested watersheds. The authors concluded that their extensive survey across several vegetation and soil types found no cases where the increase in nitrate concentrations were as marked as those found after harvesting northern hardwoods forests in the White Mountains of New Hampshire.

Leading Ridge Experimental Watershed, Pennsylvania

Three watersheds, ranging in size from 43 to 124 ha, were gauged in 1973 to begin studies on the effects of clearcutting on water quality (Lynch et al. 1975; Lynch and Corbett 1990). The watersheds have southeastern aspects on slopes of about 15% at elevations of 240-440 m. Soils average about 1.7 m in depth and are classed as Ultisols (on highly weathered, residual substrates) and Inceptisols (on steeper and less developed portions of the landscape). The major portion of the watersheds are underlain by shale and sandstone, with quartzite bedrock near the ridges. The vegetation consisted of an 80-year-old forest of oaks, hickories, and maples that regenerated after harvesting.

A key focus of this research was the application of "best management practices" designed to minimize the impacts of harvesting on water quality. The practices included

1) harvesting only 43% of the watershed;
2) retaining 30 m buffer strips along streams;
3) monitoring the harvesting contractor's performance;
4) designing of main roads, skid trails, and landings by a professional forester; and
5) rehabilitating all roads and trails after logging.

Given this design, stream turbidity (fig. 15) showed no major response to the 1976 harvesting operation, aver-

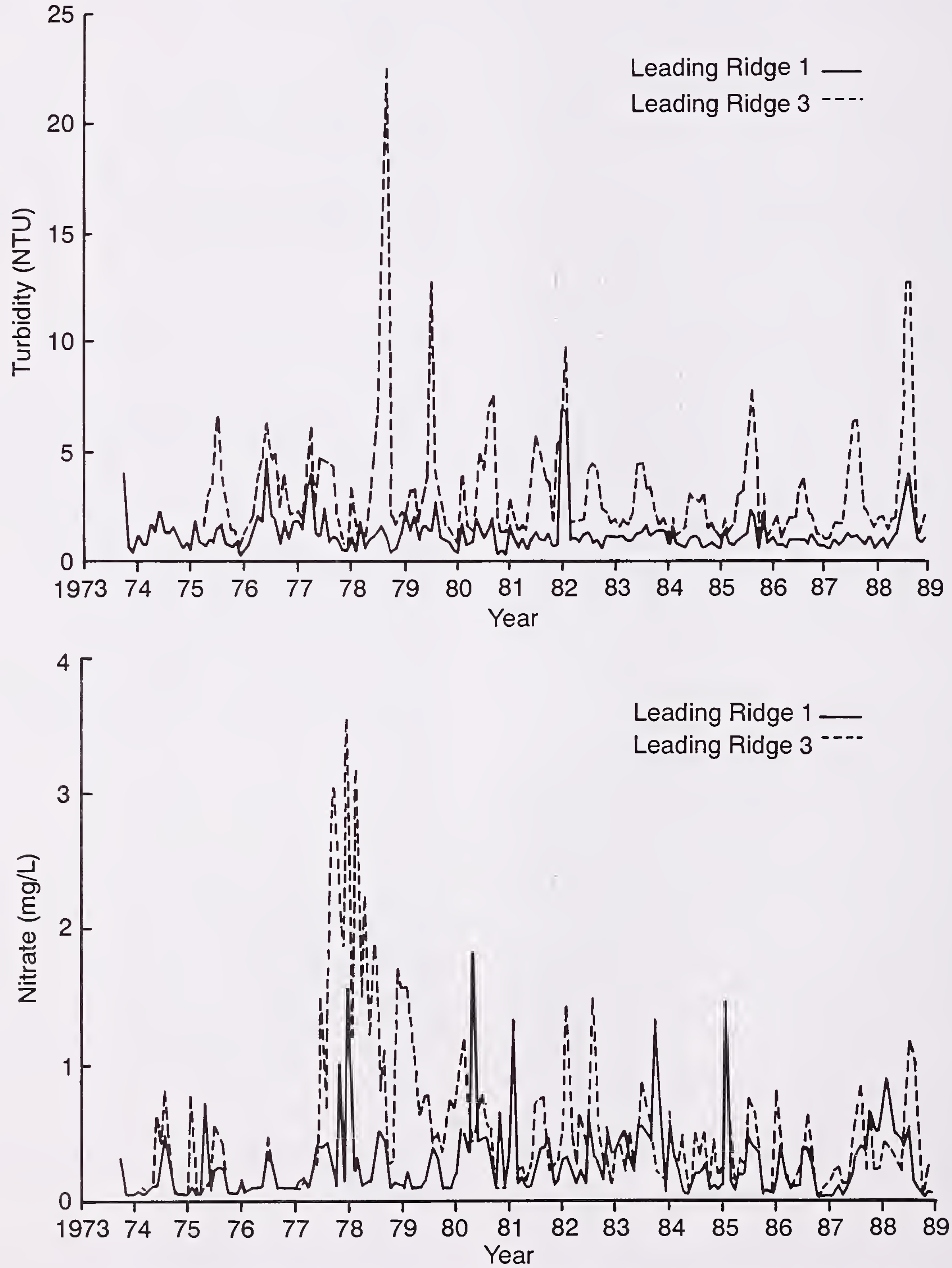


Figure 15. Trends in turbidity and nitrate concentrations for the Leading Ridge watersheds in Pennsylvania. LR1 = control, LR3 = harvested (from Lynch and Corbett 1990).

aging about 2 nephelometric units (NTU's) in the control watershed (1.7 mg of sediment/L) and 3 NTU's (5.9 mg/L) in the harvested watershed in the first year after harvest. However, turbidity peaks of up to 23 NTU's occurred in the second and third years following harvest because of disturbances in or near the stream channel. Uprooting of trees caused debris dams that caused the streams to create new channels, which increased sediment concentrations. A 450 m stretch of an intermittent stream was left with no buffer strip of trees; streamflow became perennial after harvesting, and substantial blowdown of residual trees along this stream contributed a major portion of the increased turbidity.

Stream temperatures increased by about 0.6 °C (1.6 °C maximum increase) (Rishel et al. 1982). This was attributed mostly to the stretch of the (formerly) intermittent stream that had no buffer strip. Temperatures returned to preharvest values within 2 yr, as regrowth occurred adjacent to the channels.

Nitrate-N concentrations increased substantially after harvest, reaching peak concentration of about 0.85 mg/L and an annual average of 0.08 mg/L (compared with 0.03 mg/L for the control watershed). Elevated nitrate concentrations lasted for about 5 yr, but remained well below drinking water standards. Other changes in stream chemistry in the first 2 yr included modest decreases in the concentrations of most ions as stream water pH and alkalinity declined. Alkalinity returned to near-preharvest levels in year 3 as stream pH increased relative to the control.

A third watershed was harvested and herbicides were applied to impede revegetation, for research purposes only (Lynch et al. 1985). The impacts of herbicide treatment after harvesting (two-year results) were much greater than those of harvesting alone:

- Annual average sediment load was 80 mg/L (resulting from stream slope failures).
- Stream temperature increased more than 10° C in summer.
- Annual average nitrate-N concentration was 2.5 mg/L.

Herbaceous plants grew rapidly after herbicide application stopped in the second year. Stream temperatures were about 6° C higher than the control in July, with a maximum observed increase of 10° C (Rishel et al. 1982).

Fernow Experimental Forest, West Virginia

The Fernow Experimental Forest in northcentral West Virginia is typical of much of the forested land in the central Appalachian region (Patric 1980). Elevations range from 650 to 1000 m, with slopes commonly between 20 and 30%. The soils are Typic Dystrochrepts (poorly developed horizons), and the vegetation is dominated by 40- to 50-year-old oaks and maples. Prior to 1940, chestnut comprised about one-fourth of the forest biomass but was wiped out by chestnut blight fungus. The experimental watersheds are about 35 ha.

The first harvest experiment (Watershed 3) involved selective logging of 13% of the basal area in Watershed 3 in 1958; no effects were observed on water yield, quality, or stormflow. Later harvests in 1963 (8% of basal area) and 1968 (6% of basal area) similarly had no impacts. In 1969, 91% of the watershed was clearcut, with a buffer strip retained to shade the stream. In 1972, this buffer strip was removed to examine effects of canopy opening on stream temperature.

The buffer strip prevented any change in stream temperature until 1972, when removal of the strip raised mean temperature by about 2 °C during the growing season. Other studies at Fernow have reported maximum temperature increases of 4 °C (Kochenderfer and Aubertin 1975). Harvesting had little effect on turbidity; non-stormflow turbidity was always below 2 NTU in both harvested and control watersheds, and the maximum observed turbidity (40 NTU) from stormflow occurred in the control stream. Annual average turbidity was 3.1 NTU for the harvested watershed from 1960 through 1976, compared with 2.1 NTU for the control watershed. No effects of harvesting on annual average stream water concentrations were apparent, although maximum stream water nitrate-N concentrations reached 1.4 mg/L in the harvested watershed (Aubertin and Patric 1974). The researchers concluded that forest harvesting had only minor and short-lived effects.

Commercial clearcutting of Watershed 1 (Kochenderfer and Aubertin 1975) had much greater impacts on the stream, including turbidity averages of about 500 Jackson turbidity units (JTU) and stormflow peaks of 56,000. The control watershed had an average of 2 JTU and a peak of 25 JTU. (It is not possible to convert JTU to NTU because of different techniques used [MacDonald et al. 1991].)

Several studies have examined the effects of forest fertilization on stream water quality at Fernow. Kochenderfer and Aubertin (1975) reported peak nitrate-N concentrations of 16 mg/L in October following fertilization with 225 kg-N/ha as urea after clearcutting Watershed 1. (Note that normal forest practices would not include fertilization soon after harvesting.) Helvey et al. (1989) and Edwards et al. (1991) examined the effects of fertilization with N (340 kg-N/ha as ammonium nitrate) and P (100 kg-P/ha as triple superphosphate) on stream water chemistry and nutrient yields from intact forests. Fertilizer was applied in the spring of 1976 to 2 watersheds, one with a south-facing aspect, and the other north-facing. Application was by hand (with a cyclone seeder), with no fertilizer applied directly to the streams. Stream water nitrate-N concentrations exceeded the 10 mg/L drinking water standard for

3 weeks during the autumn after application. About 23% of the fertilizer-N (80 kg/ha) was lost as nitrate-N in stream water in 3 yr after application. Phosphate concentrations showed no increase.

Summary

Watershed-level examinations of the effects of forest practices on water quality in the Northeast have focused on stream water chemistry; sedimentation problems can be controlled with best management practices (Martin and Hornbeck 1992). Nitrate-N concentrations have increased substantially in some cases in the northern hardwoods vegetation type in New Hampshire. The increased nitrate-N concentrations remained below water quality standards for normal clearcutting operations but substantially exceeded acceptable levels in the devegetation experiment at Hubbard Brook (table A.4, fig. 14). Sediment loads in streams draining watersheds with intact forests may exceed standards during storms; the response to forest harvest depends strongly on road location and design and on degree of disturbance of riparian vegetation. The use of best management practices allows forest harvest to have minimal effects on stream water quality (see Chapter 10).

North Central and Great Plains

Forests cover about 28% of the north-central states of Ohio, Michigan, Indiana, Illinois, Iowa, Wisconsin, and Minnesota; rangeland covers only 0.6% of this region (USDA Forest Service 1982). Forest productivity averages about 1.4 to 3.5 m^3/ha annually, with only 2% of forestlands producing more than 8.4 m^3/ha annual increments. About one-third of the forests are combinations of maple, beech, birch, and aspen; another third are variations on the oak/hickory type. Conifer forests such as white pine, jack pine, red pine, and white spruce constitute about 18% of the forests. About 13% of the forests are federally owned.

Precipitation grades from about 1000 mm/yr in the southeast part of the region to 500 mm/yr in the northwest part of the region. Runoff follows a similar pattern, ranging from about 400 mm in the southeast to less than 150 mm in the northwest.

Forests cover only about 9% of the Great Plains states of North and South Dakota, Nebraska, Kansas, Oklahoma, and Texas, and rangelands cover another 44% (USDA Forest Service 1982). Forest productivity is similar to the north-central region, except for higher productivity of some pine forests in southeast Texas. About 5% of the forests in Texas exceed > 8.4 m^3/ha annual increment, but only 23% of the forests north of Texas reach even 3.5 m^3/ha. The condition of rangelands in the region is generally poor; 55% are classified as in poor or very poor condition from overgrazing by livestock. Annual forage production ranges from < 1 Mg/ha for desert grassland and shrubland, to 1 Mg/ha for plains grassland, to 2.5 Mg/ha for Texas savannah, to 3.5 Mg/ha for prairie. Only 7% of the forestland and 3% of the rangeland are in federal ownership.

Precipitation shows strong gradients across the Great Plains, decreasing from east to west and from south to north. A maximum of about 1200 mm/yr falls in southeast Texas, compared with 400 mm in southwest Texas and in North Dakota. Runoff follows a similar pattern, with 200 mm/yr in east Texas, < 30 mm/yr in west Texas, and < 30 mm/yr for most of the Dakotas (runoff from the Black Hills is about 80 mm/yr).

Very little research in these two regions has focused on impacts of forest practices on water quality.

Marcell Experimental Forest, Minnesota

Verry (1972) examined the effects of clearcutting of a watershed in northcentral Minnesota. The 35 ha watershed was typical of the region, which has rolling moraines, glacial lake beds, outwash plains, and many lakes. The soils are developed in deep (> 2 m) glacial till overlying sand, and some deep percolation may occur from the watersheds. Aspen and birch trees, about 50 yr old, dominated the forests in the upland portions. Black spruce dominated the poorly drained bog that covered about 20% of the watershed. The harvesting operation cut all trees greater than 3 m in height in the upland portion. Merchantable material was removed from the site. The bog portion of the watershed was not cut.

Harvesting had virtually no effect on water quality; nitrate-N concentrations in the control watershed averaged about 0.3 mg/L and were even lower in the harvested watershed (about 0.015 mg/L). Sediment losses were not measured because they were expected to be small in this region of very low relief. Factors that may have contributed to the lack of change in water quality include leaving trees < 3 m in height; aspen suckers revegetating the site rapidly; and leaving the spruce bog intact.

Pellston, Michigan

Richardson and Lund (1975) examined the effects of clearcutting of aspen on soil solution chemistry at 3 locations in Michigan. The soils were classified as good, intermediate, and poor on the basis of aspen production. A pair of stands (1 ha in size) was chosen at each location, and 1 was commercially clearcut with the slash material left on the site. The height of the aspen in the 60-year-old stands was: 26 m good site, 21 m intermediate site, and 15 m poor site. Soil leachate was collected with tension lysimeters, and no significant response to cutting was observed. The maximum nitrate-N concentration in soil solution was 0.07 mg/L in the clearcut at the good site.

Cherokee County, Texas

One study in east Texas examined the impacts of harvesting and site preparation on stormflow and sediment yields (Blackburn et al. 1986; Blackburn and Wood 1990). Nine small watersheds (2.6 - 2.7 ha) were examined, with slopes ranging from 4-25%. Precipitation averages 1070 mm/yr and is well distributed through the year. Soils developed in marine sediment sandstone and are classified as Typic Hapludults. The vegetation was primarily shortleaf pine, mixed with various hardwoods. Three replicates of each treatment were applied: control; harvesting followed by roller chopping and broadcast burning; and harvesting followed by shearing, windrowing, and burning. Rubber-tired tractors skidded logs to landings outside the watersheds. All plots were planted to loblolly pine.

After harvest and site preparation, mineral soil was exposed on 16% of the chopped watersheds and on 57% of the shear/windrow watersheds. About 5% of the area of the chopped watersheds showed soil movement (either erosion or deposition), compared with 47% of the shear/windrow watershed area. In the first year after treatment, stormflow in the shear/windrow watersheds (146 mm) greatly exceeded that from the chopped (83 mm) or control (26 mm) watersheds. Sediment concentrations and yields in the chopped watershed were not increased relative to the control watersheds, but the shear/windrow watersheds showed 20-fold increases (to 2100 mg/L) in the first year. Sediment concentrations in the shear/windrow watersheds dropped to about twice that of the controls in years 2, 3, and 4. Sediment yields for the control and chopped watersheds averaged about 0.03 Mg/ha across the 4 years, compared with 2.9 Mg/ha lost from the shear/windrow watersheds in year 1. By year 4, sediment yields had declined in the shear/windrow watersheds to about 0.2 Mg/ha.

Prior to harvesting, stormflow concentrations of nitrate-N (0.015 mg/L) were low, but total nitrogen (0.7 mg/L) were moderate. Concentrations in the shear/windrow plots jumped to about 0.3 mg/L nitrate-N and about 0.8 mg/L total N in the first year after treatment. Nitrate concentrations declined substantially over the next 3 yr, and did not differ from the controls in year 4. The chopped watersheds showed less response, and the observed nitrate increases did not differ significantly from the values for the control watershed.

The authors concluded that the shearing/windrowing operation exposed too much of the clayey subsoil, which accounted for increased sediment loads even 4 yr after treatment. They view the chopping treatment as the best management practice and conclude this approach produces relatively low sediment losses. Nitrate concentrations were not critical in either treatment.

Grazing Impacts on Water Quality

Few studies are available on the water quality impacts of grazing in forests in this region; some information is available on soil compaction and infiltration rates, and some chemical and microbiological information is available from unforested pastures.

As an example of soil compaction problems with grazing, Stoeckler (1959) showed that infiltration rates under grazed oak woods in southwestern Wisconsin averaged only 1.2 mm/hour, compared with 31 mm/hr for ungrazed areas. Orr (1975) examined the recovery of soils that were compacted by cattle grazing in the Black Hills National Forest of South Dakota. Water quality was not assessed, but over 4 yr, removal of grazing pressure allowed substantial increases in soil infiltration rates and decreases in surface runoff rates.

Doran and Linn (1979) documented bacteriological water quality parameters for grazed and ungrazed pastures in eastern Nebraska. Bacterial concentrations generally exceeded drinking water standards in both pastures. The impact of grazing was evident: Fecal coliform concentrations showed a 5- to 10-fold increase over those in the ungrazed pastures.

Grazing impacts were also examined at the North Appalachian Experimental Watershed near Coshocton, Ohio (Chichester et al. 1979). Summer grazing systems had little impact on water chemistry, but a winter feeding system increased losses of sediments, nutrients, and organic carbon. The increases were not considered to be excessive relative to water quality standards.

A similar study at the Oklahoma South Central Agriculture Research Station near Chickasha examined the effects of a rotational grazing system with an overgrazed, continuous system (Menzel et al. 1978). Runoff from the rotational grazing system pasture averaged about 43 mm/yr, compared with 98 mm/yr for the continuously grazed pasture. Concentrations of sediment and nitrate-N averaged 700 mg/L and 0.15 mg/L for the rotational pasture, and 8,300 mg/L and 0.3 mg/L for the continuously grazed pasture.

Summary

Few studies have examined water quality in these regions (table A.5). The results from east Texas (Blackburn et al. 1986) are consistent with those from the southeastern United States—use of best management practices leads to minor effects on water quality, whereas extreme site preparation treatments may release unacceptable quantities of sediment into streams. In the north, harvest of aspen stands on relatively level terrain appears to have minimal effects, but more studies would be needed to determine the extent of this pattern beyond the two sites studied. Information on the effect of grazing on water quality is limited; overgrazing does appear to degrade water quality.

Chapter 6

Rocky Mountains

About 25% of the Rocky Mountain states (Montana, Idaho, Wyoming, Colorado, Utah, and Nevada) is forested, with rangelands accounting for an additional 60% (USDA Forest Service 1982). Federal lands contain 75% of the forests and 65% of the rangelands. The productivity of forests in this region is generally low, with over 40% of the forests producing less than 1.4 m^3/ha of wood annually. Dominant forests in the region include mixed woodlands of pinyon pine and junipers (20%), lodgepole pine (17%), mixed forests of spruce and fir (16%), Douglas fir (16%), and ponderosa pine (8%). Rangeland condition is generally fair to very poor because of overgrazing; only 15% is in good condition.

The region as a whole is semiarid, with precipitation averaging less than 500 mm/yr in the plains. Precipitation is greater in the mountains, reaching about 900 mm/yr in parts of the Colorado Rockies and 1250 mm/yr in parts of the Idaho/Montana Rockies. Average annual runoff ranges from less than 30 mm/yr for the plains, to 500 mm/yr in the Colorado Rockies, to over 1000 mm/yr in parts of Idaho and Montana. Most harvesting operations in these regions involve removal of natural, old-growth forests, followed by natural regeneration.

Fraser Experimental Forest, Colorado

The Fraser Experimental Forest was established in 1937 for timber and watershed research (Alexander et al. 1985; Alexander 1987). The 9300 ha research forest is about 50 km west of Denver, and it ranges in elevation from 2680 to 3900 m. Two watersheds, Fool Creek and Deadhorse Creek, have been the focus of intensive research on the impacts of forest harvest patterns on snow accumulation and water yield.

Geography and vegetation

The topography is typical of the southern Rocky Mountains. The west side of the forest is rugged, with steep-sided valleys filled with glacial alluvium. The south and east sides are remnants of an old peneplain, characterized by long, gentle slopes. Soil parent materials are derived from gneiss and schist, with small outcroppings of granitic rocks. Small areas of sandstone are also present at high elevations in the west side of the forest. The majority of the soils are Dystrochrepts. Some Spodosols occur.

Lower elevation forests (below 3000 m) are largely dominated by lodgepole pine (depending on aspect). Occasional patches of aspen occur in wetter microsites. Forests of Engelmann spruce and subalpine fir dominate at middle elevations, and treeline is at about 3300 m. Douglas fir occurs in both lodgepole pine and spruce/fir stands. Aboveground biomass in a 60-year-old lodgepole pine stand is on the order of 140 Mg/ha, compared with 160 Mg/ha for a 230-year-old stand (Ryan and Waring 1992).

Climate

The climate is cool, with long, cold winters and short, cool summers. The average yearly temperature at the headquarters building at 2740 m elevation is about 1 °C; mean monthly temperatures for January and July are -10 °C and 13 °C. Precipitation over the entire forest averages about 880 mm/yr and is evenly distributed through the year (fig. 16). About two-thirds of precipitation falls as snow between October and May.

Hydrology

Snowmelt contributes 90% or more of the annual streamflow (Troendle and Kaufmann 1987). Fool Creek is gauged both at the lower alpine limit (above 3200 m) and at the bottom of the watershed (2900 m). Snowmelt occurs earlier in the alpine than in the lower part of the watershed, with peak snowmelt flows differing by about 2 weeks (fig. 17). The alpine area comprises about 23% of the watershed but contributes about 34% of streamflow. On a unit area basis, water yield from the alpine portion of the watershed is about 80% greater than the lower portion. Clearcutting increased streamflow by about 200 mm/yr (60%) for the first 3 yr after harvest. Most (150 mm) of the increased flow results from increased water content of snowpack accumulated in spring; reduced evapotranspiration contributes only about 50 mm/yr. Greater snowpack accumulation results from reduced interception loss of snow suspended in canopies; sublimation rates are very high for snow suspended in canopies under the dry, windy conditions in winter.

Baseline water quality

Precipitation at Fraser (Stottlemyer and Troendle 1987) is a very weak acid solution (pH about 5.3) comprising nitrate (20% of anions), sulfate (35% of anions), chloride (24% of anions) and bicarbonate (21%). Calcium and sodium dominate the cations, with H^+ accounting for only 10% of the cation charge. Bicarbonate (alkalinity) contributes most (85%) of the anions in stream water draining East St. Louis Creek (the control watershed for Fool Creek); sulfate and chloride each

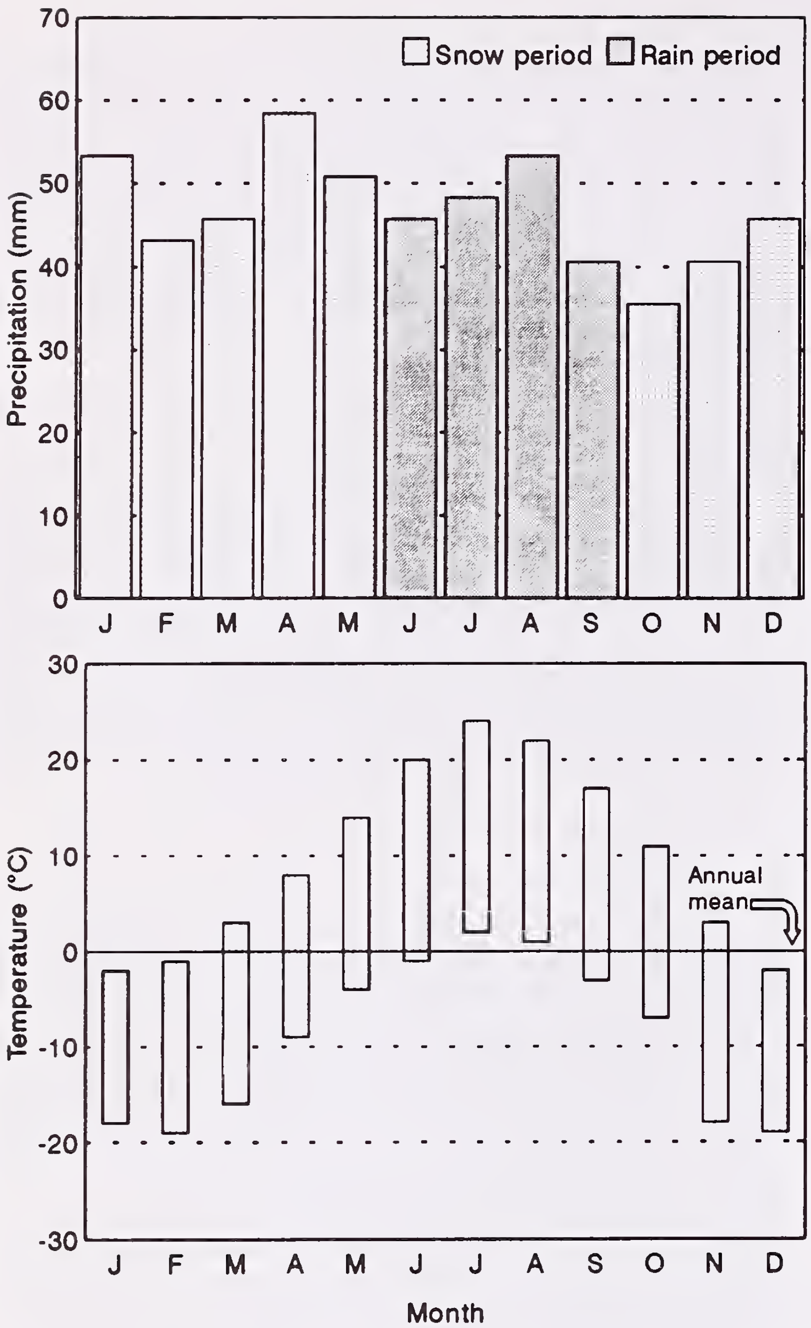


Figure 16. Average precipitation and temperature (bars represent average daily range) for the Fraser Experimental Forest, Colorado (from Alexander et al. 1985).

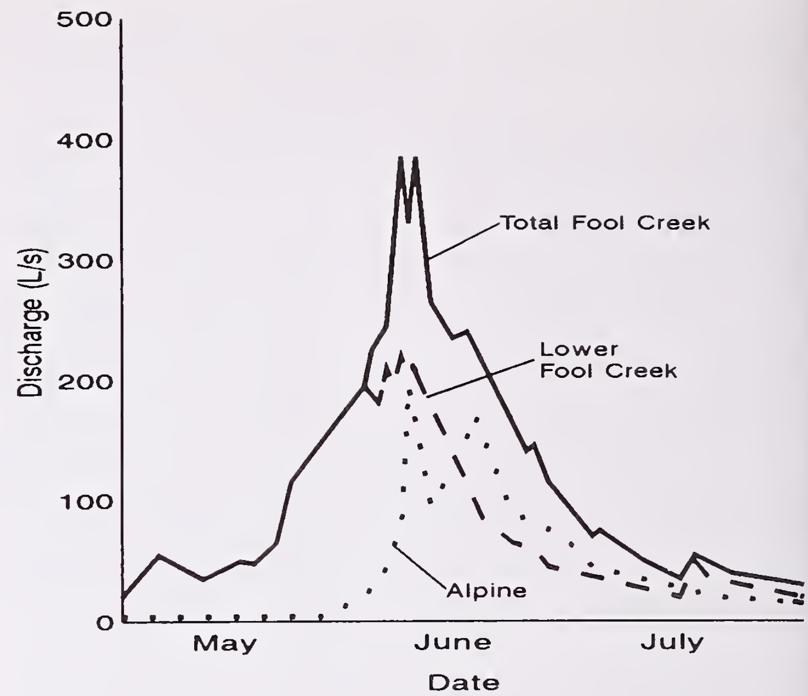


Figure 17. Hydrograph for the entire Fool Creek watershed alpine, and lower portions of Fool Creek (from Alexander et al. 1985).

contribute about 5%; and nitrate-N concentrations are very low (about 0.04 mg/L). Calcium dominates the cations (60%), with magnesium, sodium, and potassium comprising most of the rest.

Water quality responses to treatments

Fool Creek. The Fool Creek watershed (290 ha) contained about 220 ha of merchantable forest in 1950, most of it "overmature" stands of 250- to 350-year-old lodgepole pine, Engelmann spruce, and subalpine fir (Alexander et al. 1985; Alexander 1987). As part of the harvesting experiment, 5 km of main roads and 14 km of spur roads were constructed between 1950 and 1952. Spur roads were spaced about 180 m apart along the contours. Timber harvesting between 1954 and 1956 removed trees in alternating strips perpendicular to the contour; the strips ranged from 20 to 120 m in width. No trees were cut within 25 m of the streams. All stems >10 cm were cut, across an area of 100 ha, and roads covered another 15 ha. After harvesting was completed, spur roads were seeded with grass and culverts were removed from half of the spur roads. The main road was routinely maintained, including grading and graveling. Sediment yields increased by about 10- to 20-fold after harvesting to about 0.2 Mg/ha during the 3 yr of road construction and harvesting. After about 5 yr, sediment yields declined to about 0.05 Mg/ha (2 to 4 times that of the unroaded control watershed). During the peak stormflows in 1964 and 1965, suspended sediment concentrations remained below 5 mg/L.

Deadhorse Creek. About 14 km of roads were built in the 270 ha Deadhorse Creek watershed over a period of 26 yr. Portions of the watershed were harvested between 1977 and 1984. The North Fork received 1 ha patchcuts on one-third of its 40 ha. About 35% of the trees >180 mm diameter at breast height were individually cut from the 40 ha North Slope unit. About one-third of the forested area (one-fourth of total 37-ha area, including alpine) of the Upper Basin Unit was harvested in irregularly shaped patches that ranged in size from 0.4 to 2 ha. Sediment yields were smaller than for the harvesting of Fools Creek, remaining below 0.06 Mg/ha annually in all years. Stream water nitrate-N concentrations increased by about 3-fold for the entire watershed (for 2 yr during harvesting and 2 yr after harvesting) to about 0.02 mg/L; concentrations for the North Fork reached 0.06 mg/L (Stottlemyer 1987).

Subsurface flow study. Troendle and Nilles (1987) examined the effects on subsurface waterflow and chemistry of harvesting a 0.4 ha plot in an old-growth stand of lodgepole pine, Engelmann spruce, and subalpine fir. Nitrate-N concentrations for water flowing in the upper 1 m of the soil mantle from the uncut slopes averaged about 0.04 mg/L; concentrations reached 0.015 mg/L for the 1-4 m depth. One and a half years after harvest, peak nitrate-N concentrations reached 1 mg/L for solutions from the deeper portion of the profile. The increases were dramatic, but the maximum concentrations were still low.

Overall, the low clay content of the soil and deep forest floor combine to keep the production of sediment much lower at Fraser than for sites reported in the South (Chapter 3). Although nitrate concentrations increased after harvesting, the post-harvest concentrations remained very low.

Silver Creek Study Area, Idaho

A large portion of the forests in central Idaho occur on the Idaho Batholith, an area with steep slopes and very erodible soils overlying highly decomposed granite (Clayton and Kennedy 1985). Construction of logging roads in the past led to gullying and ditch erosion, which led to the extensive use of helicopters for removing logs from cutting units. Clayton and Kennedy (1985) examined the water quality effects of clearcutting 23% of a 163 ha watershed. The coarse-textured soil (dominantly Typic Xerorthents) lacked cohesion and had a high potential for erosion after harvest. The mixed forest of ponderosa pine and Douglas fir had an aboveground biomass of about 150 Mg/ha. Uncut buffer strips were retained about 15 m on both sides of first- and second-order stream channels; a 30 m buffer was retained along the main stream (third order). Slash was lopped, scattered, and broadcast burned. The burn was incomplete, leaving 22% of the harvest areas unburned. Erosion rates rose substantially after harvesting to a peak of 13 Mg/ha in the first summer after harvest (rates declined to 4 Mg/ha for the second year after harvest). However, sediment loads at the weir showed much less response because of retention of sediments on slopes and within the channel. Sediment storage in the main stream channel equals about 12 yr of annual sediment flux. Nitrate-N losses in stream water increased about 10-fold after harvest, but this represented a maximum loss of just 0.1 kg/ha annually and a peak nitrate-N concentration of 0.06 mg/L. Nitrate concentrations returned to control-levels within 5 yr.

Farther north in the Clearwater National Forest (another portion of the Idaho Batholith), a total of 871 landslides over a period of 3 yr moved about 183,000 m^3 of soil, with about 23% of it reaching streams (Megahan et al. 1978). Roads, or roads plus logging, were associated with almost 90% of the slides.

Priest River Experimental Forest, Idaho

Three sites were examined in the Priest River Experimental Forest for the effects of clearcutting and slashburning on water quality (Snyder et al. 1975). Elevation of the three sites spanned from 745 to 1220 m. Precipitation totals about 900 mm/yr, with snow comprising most of the total. Summers are dry. Bedrock is mostly Precambrian metamorphic rock, with a thick cap of volcanic ash. The dominant tree species include western white pine, western redcedar, Douglas fir, and western larch.

The Benton Creek unit was a 44-ha clearcut along a sideslope of Benton Creek, with a 60 m uncut buffer strip separating the unit from the stream. The forest was dominated by Douglas fir and western white pine, which were about 100 yr old. Three sampling locations were established on Benton Creek: one above, one just downslope from the unit where an ephemeral stream enters Benton Creek, and one below. The middle sampling point showed greater concentrations of sediments (4.5 mg/L above the unit, 6.4 mg/L on the unit) and nitrate-N (0.17 mg/L above, 1.5 mg/L on). However, the lower sampling point showed values comparable to the upper station, indicating dilution or removal of sediment and nitrate within the stream.

The Ida Creek unit was a 2.6 ha clearcut in the headwaters of Ida Creek with nearly level topography. The forest was dominated by western larch, Douglas fir, and western white pine. Logging slash was windrowed and burned. A 60-m buffer strip separated the unit from Ida Creek. Sediment concentrations were significantly greater adjacent to the unit (37 mg/L) than above (7.1 mg/L) or below (12.0 mg/L) the unit, but no effects of harvesting were apparent on nitrate-N concentrations (about 0.15 mg/L).

The Canyon Creek unit contained a 23 ha clearcut and a 3 ha partial cut of a set of stands averaging about 100 yr old (white pine, redcedar, larch, Douglas fir, spruce, and lodgepole pine). The unit was separated from Canyon Creek by a 30 m buffer strip. The sample point adjacent to the unit showed increased sediment concentrations (16 mg/L) compared to upper (3 mg/L) and lower (4 mg/L) stations. Nitrate-N was also higher in the adjacent station (0.18 mg/L) than in the upper and lower stations (about 0.015 μg/L).

Coram Experimental Forest, Montana

Stark (1979) examined the effects of harvesting and site preparation on soil solution chemistry in a larch/Douglas fir forest. The study site was on a steep slope (35-45%), ranging in elevation from 1000 to 1900 m. Precipitation at lower elevations averages about 790 mm/yr, most falling as snow. Mean annual temperature is about 6 °C. Soils (mostly Cryoboralfs) formed in a variety of parent materials, including shale, limestone, and quartzite; texture is very coarse, with about 50% rock content by volume. Harvest treatments included two replicates of the following: shelterwood, group selection (very small clearcuts), and clearcutting. Soil solutions were collected at 1 m depth by using ceramic cup tension lysimeters. Prior to logging, soil solution concentrations of nitrate-N were on the order of 0.4 to 1.2 mg/L. Nitrate-N concentrations rose in the first year after harvesting and site treatments, to about 1.5 mg/L for the shelterwood cuts, to 3.6 mg/L for the clearcut with low-intensity burn, and to a maximum of 6.4 mg/L for the clearcut and intense burn plots.

This study identifies a potential opportunity for harvesting and site preparation treatments to increase stream water nitrate concentrations, but the study was not designed at a watershed scale. It is also quite possible that elevated nitrate concentrations would have been greatly reduced as the water from the harvested units percolated through undisturbed forests.

Bitterroot National Forest, Montana

An unpublished master's thesis (Bateridge 1974, cited in USEPA 1980) examined 3 harvested and 3 uncut control watersheds in the Bitterroot National Forest in western Montana. The watersheds ranged in size from 41 to 350 ha and contained forests of mixed conifers, including ponderosa pine, Douglas fir, lodgepole pine, Engelmann spruce, and subalpine fir. Three years after harvesting, Lodgepole Creek had average nitrate-N concentrations of 0.19 mg/L, compared with the control Spruce Creek with 0.11 mg/L. The first year after harvesting and tractor-piling of slash, nitrate-N concentrations averaged 0.17 mg/L in Mink Creek compared with 0.13 mg/L for the control Springer Creek. In the first year following tractor-piling and burning of logging slash, Little Mink Creek averaged 0.4 mg/L of nitrate N compared with 0.17 mg/L for the control. These comparisons indicate slight increases in nitrate concentrations following harvesting and site preparation, but the responses were too low to be important.

Alberta, Canada

Singh and Kalra (1975, cited in USEPA 1980) examined water quality in 13 watersheds in west-central Alberta, examining effects of harvesting of lodgepole pine, white spruce, and aspen forests. Water sampling was restricted to spring snowmelt and summer recession periods. All nitrate-N concentrations were very low, ranging between 0.005 and 0.05 mg/L, with no apparent differences between harvested and control watersheds.

Union Pass, Wyoming

The Union Pass area in western Wyoming is a gently rolling, glaciated plateau at about 3000 m elevation. The soils (mostly Mollic Cryoboralfs and Mollic Cryoborepts) are moderately deep (> 1 m), high in rock content (> 50% by volume below 0.5 m), and mottled below 1 m depth (indicating periods of water saturation and anaerobic conditions). The forest was mostly (> 75%) lodgepole pine, with some Engelmann spruce, subalpine fir, and limber pine (DeByle 1980). Four 8-ha units were clearcut: two by conventional methods (tree length boles skidded with crawler tractors), and two by whole-tree methods (whole trees skidded with rubber-tired skidders; logs separated and remaining biomass chipped and removed). The conventionally harvested units received two site preparation treatments (with 2 replicates/unit): broadcast burning or windrowing and burning. Little debris was left on the whole-tree harvest units, so the site preparation treatments involved minimal amounts of debris, or application of chipped debris to an average depth of 0.1 m. Soil solutions were obtained from ceramic cup tension lysimeters at 0.6 and 1.2 m depths. Nitrate-N concentrations (apparently averaged across both depths?) averaged about 0.05 mg/L for the undisturbed forest and were increased by 2- to 100-fold in treated units (over a period of 3 to 6 yr). The maximum annual average was 4.4 mg/L beneath burned windrows (1.2 mg/L for spaces between windrows). Eight of the samples from beneath windrows exceeded 10 mg NO_3-N/L. The study was not designed to address stream water quality, but DeByle (1980) noted that considerable dilution of the high nitrate levels would occur before soil water reached a stream.

Chicken Creek, Utah

Johnston (1984) examined the effects of harvesting five small patches (1.2 to 4.1 ha, total of 11 ha) of aspen comprising 13% of the area of a small watershed near Farmington, Utah (about 22 km northeast of Salt Lake City). Both the control and harvest watersheds contained beaver colonies, which may have influenced stream water quality. The watersheds receive about 1140 mm/yr of precipitation, with about 80% falling as snow between November and April. Slopes are moderate, averaging 12-45%. All stems > 5 cm were cut, firewood was removed (by hand or with horses), and slash was scattered. No cutting occurred within 140 m of the stream. Harvesting produced no change in streamflow quantity, but concentrations of nitrate were increased from about 0.008 mg/L to 0.025 mg/L in the second year after cutting.

Grazing Impacts on Water Quality

It is difficult to assess the impacts of cattle grazing on water quality, because most suitable lands have been grazed with varying intensities for decades. Most studies in the region have focused on features relative to water quality (such as soil infiltration rates; Gifford 1981), rather than on water quality per se. The available studies have generally documented differences in water quality that can be attributed to recent or current differences in grazing intensity between sites. But these studies cannot quantify the long-term impacts. Furthermore, most studies have ignored the specific effects of grazing within riparian areas.

A variety of grazing experiments were conducted at the Manitou Experimental Forest near Colorado Springs, Colorado. Most studies focused on grazing impacts on small plots (40 m^2) of ponderosa pine/grassland communities. Heavy grazing for 12 yr increased runoff (from 0.3 mm/season for control plots to 8.6 mm/season for heavily grazed plots), and it increased sediment yield (0.15 Mg/ha annually for control, and 0.35 Mg/ha for heavily grazed) (Dunford 1954).

Johnson et al. (1978) examined the effects of grazing on a 2.6 km stretch of Trout Creek, which flows through the Manitou Experimental Forest. The site receives about 400 mm/yr precipitation at an elevation of 2360 m, and the streamflow declines from peaks of about 0.3 m^3/second in the spring to 0.01 m^3/second in late summer. The stream was divided into two pastures of about 100 ha each. Both pastures had been grazed in the past, primarily as holding pastures in the spring and early summer before cattle were moved to higher pastures. The upper pasture was left ungrazed in this experiment, and the lower was grazed by 75 cows with calves. The floodplain alluvial soils are derived from Pikes Peak granite; unstable stream banks were up to 2 m tall above the stream. Vegetation in the pastures was a ponderosa pine/bunchgrass community, with riparian willows and sedges.

Before the experiment began, 30 beavers were removed from the control pasture, and 7 major dams were breached. This exposed extensive silt flats in the former ponds, which contributed substantial sediment to the stream during the following season of study. Sediment concentrations were high, ranging from 35 to 65 mg/L for the grazed period (2 weeks) and dropping to 8-10 mg/L after grazing stopped. Most of the sediment was thought to result from the destruction of the beaver dams, so no conclusions were possible about the current grazing impacts. Nitrate-N showed no response (about 0.07 mg/L in the grazed period for both pastures and 0.03 mg/L after grazing). Fecal coliforms averaged 200 colonies/L for the ungrazed pasture during the high flow period, compared with 1050 colonies/L for the high flow period in the grazed pasture. Concentrations dropped to 210 and 440 colonies/L after grazing stopped in the ungrazed and grazed pastures. Fecal streptococcus counts were 73 colonies/100 mL for the ungrazed, compared with 176 colonies/100 mL for the grazed pasture.

Meiman and Kunkle (1967) compared two streams in the Front Range of Colorado: Little Beaver Creek was ungrazed, and the lower half of the Pennock Creek watershed was grazed. Both watersheds are about 18 km^2, ranging in elevation from about 2400 to 3300 m. Forests are comprised of lodgepole pine, blue spruce, Engelmann spruce, subalpine fir, and aspen. Suspended sediment concentrations were similar between the two streams, averaging 7-25 mg/L for the control (Little Beaver Creek) watershed, and 4-21 mg/L for the grazed watershed (Pennock Creek). Peak concentrations were 180 mg/L for the control and 210 mg/L for the grazed. Bacterial concentrations were greater in the grazed watershed; in 1965, total coliform counts reached 120 colonies/100 mL while the control had 37 colonies/100 mL. Fecal coliform concentrations were 4 colonies/100 mL for the control watershed, and 68 colonies/100 mL for the grazed; fecal streptococcus counts were 14 colonies/100 mL for the control and 24 colonies/100 mL for the grazed.

Buckhouse and Gifford (1976) examined the effects of grazing on water quality in the Coyote Flats study site in southeastern Utah. A pinyon/juniper woodland was chained (trees toppled by a chain pulled between two tractors) and seeded with crested wheatgrass. Grazing was excluded for 7 yr, and then the pasture was grazed for 2 weeks at a rate of 0.5 animal unit months (AUM)/ha. Artificial rainfall was applied to 6 0.23 m^2 small plots in the chained area; 6 in the chained, ungrazed area; and 6 in the unchained adjacent woodland. The water fell at a rate of 70 mm/hr for 28 minutes, with surface runoff collected at 5-minute intervals. No differences in fecal coliform concentrations were observed between grazed and ungrazed (woodland or chained) areas. Manure covered about 0.2% of the pasture, and the low density of fecal material was thought to account for the lack of grazing effect on runoff water quality. A second artificial rain experiment focused on the pattern of fecal coliform concentrations with distance from individual cowpies. Concentrations were about 700,000 fecal coliform/100 mL in the middle of the cowpies, dropping to 30,000 at 0.4 m away and 23 at 1 m distance. Counting the area of influence of each cowpie, the points of pollution occupied about 5% of the pasture;

any effect of grazing on water quality would derive from feces deposited near or in the stream.

Skinner et al. (1974) examined concentrations of bacteria in subwatersheds of the Nash Fork Watershed Study Area in the Medicine Bow Mountains in southeastern Wyoming. Concentrations of fecal coliform in a low-use, natural watershed averaged about 0.2 to 1.2 colonies/100 mL, compared with 20 to 30 colonies/100 mL for watersheds receiving intensive grazing and recreational use. They concluded that these levels represented no water quality problems relative to recreation use.

The impacts of grazing systems on water quality in sagebrush rangelands were investigated at the Reynolds Creek Watershed, about 90 km southwest of Boise, Idaho (Stephenson and Street 1978). Fecal coliform counts increased when sheep or cattle were introduced into pastures. Counts remained elevated for up to 3 months after grazing stopped. Maximum fecal coliform counts were about 2500 colonies/100 mL; the authors concluded that grazing is likely to lead to intermittent concentrations of bacteria that exceed water quality standards.

Exclusion of grazing pressures typically allows riparian vegetation to recover dramatically, often within 5 yr, and improved riparian vegetation leads to improved water quality and fisheries habitat (Schulz and Leininger 1990). At present, best management practices (BMP's) guidelines have not been firmly established for grazing impacts on water quality (W. Leininger, Colorado State University, personal communication). One challenge to the development of BMP's is variation in animal behavior among areas. The species composition of riparian areas varies dramatically across landscapes, so the water-quality impacts of a given grazing intensity within a management unit would also vary. Greater attention to the development and implementation of grazing BMP's is warranted.

Summary

The available information from small watershed studies show no general water quality problems associated with forest practices in the Rocky Mountain region (table A.6). Severe sediment problems in the granitic batholith of Idaho are an exception; excessive sedimentation of tributaries in the South Fork of the Salmon River led to a harvesting moratorium in the 1960's. Sediment concentrations are generally low across the region (unless mass movements or road-related erosion occur), and nitrate concentrations are uniformly low. Grazing may lead to excessively high bacterial concentrations in some cases.

Pacific Northwest

About 50% of Oregon and Washington is forested, with rangelands comprising another 28% (USDA Forest Service 1982). Forty-seven percent of the forests and 51% of the rangelands are federally owned. Some of the land in this region is very productive, with about 30% of the forests yielding more than 8.4 m^3/ha annually. Only 22% of the forestlands produce less than 3.5 m^3/ha annually. Douglas fir forests cover about one-third of the forest land. Other major forest types include ponderosa pine (15% of forestland), spruce/fir (15%), hemlock/Sitka spruce (10%), lodgepole pine (6%), and pinyon/juniper (5%). Because of poor grazing management, only 20% of the rangeland in Oregon and Washington is in good condition and 45% is in poor or very poor condition (USDA Forest Service 1982).

Annual precipitation spans an order of magnitude from less than 250 mm/yr in parts of the high plains of eastern Washington and Oregon to over 3500 mm/yr along the coastal mountains. In the Cascade Mountains, precipitation ranges from 1500 to 2500 mm/yr. Water yield spans a similarly large range, from near 0 in the driest eastern parts to over 2500 mm/yr near the coast.

Precipitation along Alaska's southeast coast exceeds 2500 mm/yr, declining to about 500 mm/yr or less in the interior forests and tundra. Forests cover about one-third of Alaska; most of the rest of the state is classified as "rangelands," consisting mostly of open tundra ecosystems (USDA Forest Service 1982). Ninety-four percent of the forestland is federally owned, compared with 62% of the "rangeland." Forest productivity is very low over most of the state, with 75% of the forestland yielding less than 1.4 m^3/ha. However, some coastal forests are very productive, and about 1% of Alaska's forests yield more than 8.4 m^3/ha annually. Most (70%) of Alaska's forests are dominated by white or black spruce, with coastal forests of Sitka spruce and western hemlock comprising about 11%. Hardwood forests (poplar, birch, and alder) account for 19% of the forested area.

H.J. Andrews Experimental Forest, Oregon

The H.J. Andrews Experimental Forest was established in 1948 and has become one of the most intensively studied forests in the world (Swanson et al. undated). The 6400 ha forest is located in the Cascade Mountains about 80 km east of Eugene, Oregon. Elevations range from 410 to 1630 m, with average stream slopes of about 45% and hillslopes of 45 to 120% (Sollins et al. 1980). About 45% of the forest is old-growth, with dominant trees over 400 yr old. At lower elevations, forests are dominated by Douglas fir, western hemlock, and western red cedar. Pacific silver fir replaces western hemlock as elevation increases, and Douglas fir and western red cedar decline in importance. Upper elevation forests consist of firs (Pacific silver fir, noble fir) and mountain hemlock. About one-third of the forest has been logged. Research in the 1950's focused on logging and regeneration, then shifted to watershed hydrology studies in the 1960's and to ecosystem studies in the 1970's. Since 1977, the site has been jointly administered by the USDA Forest Service and Oregon State University.

Landscapes at the H.J. Andrews Experimental Forest have been shaped by glacial, fluvial, and hillslope movement processes (Swanson and James 1975). The geomorphology across the forest is complex. Below about 850 m, bedrock is composed of a variety of hydrothermally altered volcaniclastic rocks. Bedrock outcrops up to about 1220 m include ash flows and basalt flow, and ridge crests are formed by andesitic lava flows. Soils developed on these parent materials are generally loamy in texture, with very high pore space (mostly Dystrochrepts). Infiltration rates for the soils are much greater than rainfall intensities (precluding overland flow). Soil water storage capacity is high in the upper 1.2 m, which holds about 0.35 m of water at field capacity. Mass movement events (such as landslides) are very important in the forest, with about 25% of the landscape covered by these features in the lower half of the forest.

Precipitation is about 2300 mm/yr, with deep snowpacks common above 1000 m. Summer precipitation is very low, averaging less than 20 mm in July and in August (fig. 18). Streamflow tracks precipitation closely (fig. 18), with a total annual streamflow of about 1510

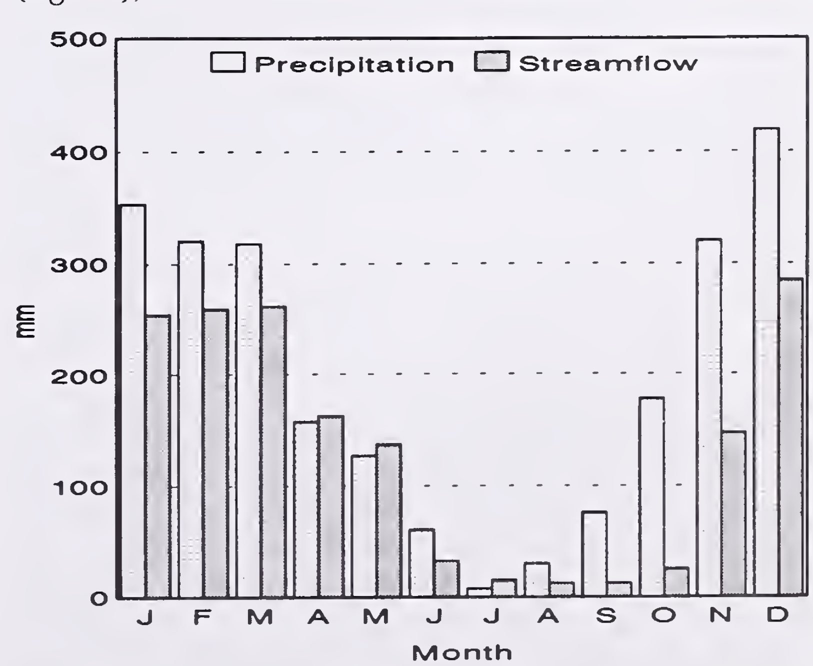


Figure 18. Average monthly precipitation and streamflow for the H.J. Andrews Experimental Forest (from Anderson et al. 1976).

mm/yr. Mean air temperature for Watershed 10 averages about 8 °C.

Baseline water quality

Precipitation chemistry (Sollins et al. 1980) is dominated by chloride (85% of total anion charge) and a combination of sodium (35% of cations), calcium (25%), and H^+ (20%). The chemistry of stream water differs markedly from precipitation, primarily from the increase in alkalinity (10 times greater than in precipitation) and similar increases in base cation concentrations. Nitrate-N concentrations in stream water flowing from Watershed 10 (prior to harvesting the old-growth forest) averaged about 0.015 mg/L, compared with <0.002 mg/L in Watersheds 6, 7, and 8 (Martin and Harr 1989).

The contribution of mass movement processes on hillslopes and within channels has been intensively characterized (Swanson et al. undated). Mass movement transports about 0.5 Mg/ha to streams annually. Inorganic material comprises 90% of the transported material, and 10% of the transported material is organic. Rare events that occur on the order of once in 300 yr contribute such large quantities of materials that over time, annual processes and rare events contribute roughly equal amounts of material to streams.

Water quality responses to treatments

Watershed 1 (96 ha) was logged between 1962 and 1966 by using a skyline system to remove logs; no roads were constructed in the watershed. Slash was broadcast burned in a "hot" fire in 1966 (Fredriksen 1970; Fredriksen et al. 1975). Logging of Watershed 1 increased streamflows by about 500 mm/yr (about a 40% increase). In the first 12 yr after logging, total sediment yield was 12 times greater than in the control watershed. Relative to the control watershed, these greater yields included greatly increased sediment concentrations during stormflows (fig. 19). The yields also included sediment from materials from 7 debris avalanches (> 75 m^3 each). Sediment concentrations in the control watershed exceeded 10 mg/L for 9 days and reached a maximum of about 36 mg/L for 3 hours during the winter of 1966-1967 (Freriksen et al. 1975). In clearcut Watershed 1, sediment concentrations exceeded 10 mg/L for 96 days (almost the entire winter), exceeded 100 mg/L on 26 days, and topped 1000 mg/L on 2 days. Streamflow was generally non-turbid during the non-winter, low-flow months. Maximum summertime stream temperatures increased by almost 8 °C by the logging and burning operations (Levno and Rothacher 1969). Nitrate-N concentrations in stream water were also greatly increased by harvesting and burning, reaching a maximum concentration of about 0.4 mg/L (fig. 20) (Fredriksen and Harr 1979).

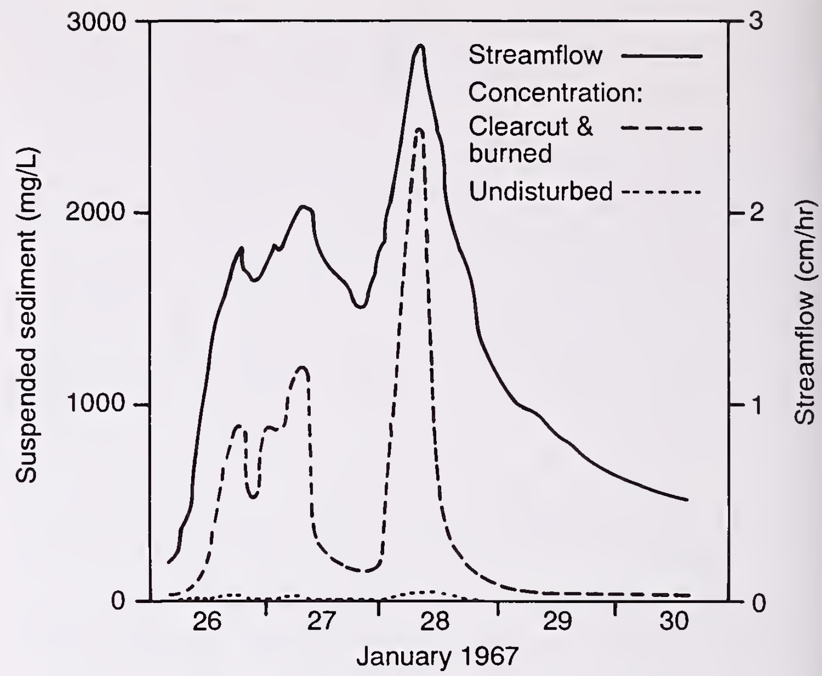


Figure 19. Storm hydrograph and suspended sediment for control and harvested watersheds at the H.J. Andrews Experimental Forest (from Fredriksen and Harr 1979).

Watershed 3 (101 ha) received 3 clearcuts (from 5 to 11 ha) on 25% of the watershed area in 1963, and logs were yarded with a high-lead cable system (Fredriksen et al. 1975). Slash was broadcast burned. About 2.7 km of roads were built within the watershed in 1959. Road construction increased sediment concentrations from about 3 mg/L in the control watershed to about 15 mg/L in the first 2 yr. Average concentrations during peak flows were increased somewhat by roads. In the third year after road construction, Watershed 3 averaged 260 mg/L for the entire year, reaching maximum concentrations of over 6300 mg/L. Clearcutting and burning 25% of the watershed in 1963 further increased stream sedimentation. The maximum increases again occurred in the third year after harvest (2500 mg/L annual average, > 15,000 mg/L maximum). Increases probably resulted from road failures rather than failures elsewhere in the watershed (R. Beschta, Oregon State University, personal communication).

Watersheds 6 (13 ha) and 7 (15 ha) are underlain by welded and non-welded ash flows in portions of the watershed, and by basalt and andesite lava flows (Martin and Harr 1989). Slopes range between 20 and 40%, which are more moderate than Watershed 1, 3, or 10. Soils are predominantly Typic Haplorthods. The forests contained 130-year-old Douglas fir forests, with some scattered Douglas fir trees of about 450 yr old. A permanent, all-weather road was built in Watershed 6 in 1964, and the forest was clearcut in 1974. Logs were yarded uphill with a high-lead cable system, and the watershed was broadcast burned in the spring of 1975. About 60% of the total basal area of Watershed 7 was harvested in 1974 in the first cut of a shelterwood

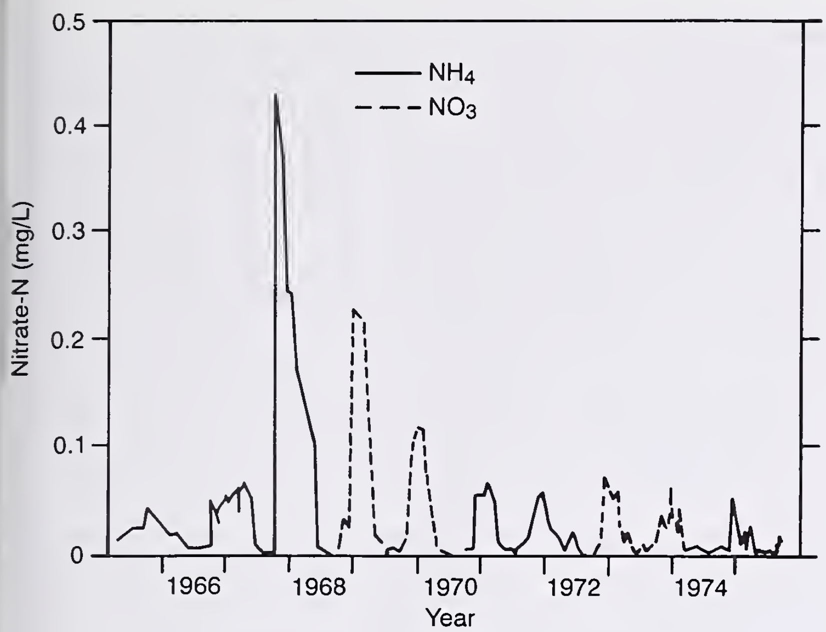


Figure 20. Concentrations of nitrate-N in streamflow following clearcutting and burning at the H.J. Andrews Experimental Forest (watersheds 1, 2 and 3, from Fredriksen and Harr 1979). Dashed lines are estimated trends for unsampled periods.

regeneration system. Logs in the upper half of the watershed were yarded by tractor, and logs in the lower half were partially suspended during yarding by a skyline cable system. Logging residue was broadcast burned only on the lower half of the watershed.

Mean annual sediment concentrations were not affected by treatments in either watershed for the first 10 yr after harvest. Nitrate-N concentrations averaged less than 0.002 mg/L before harvesting and increased 6-fold in the shelterwood system watershed and 20-fold in the clearcut watershed. Even in the clearcut watershed, the concentration of nitrate remained extremely low (less than half the concentration found in precipitation).

Watershed 10 (10.2 ha) was the principal study site for the Coniferous Forest Biome of the International Biological Program during the 1960's and 1970's. The watershed was clearcut in 1975, with logs removed by a skyline yarding system (Sollins and McCorison 1981). Woody residue > 0.2 m in diameter or longer than 2.4 m was hauled to the landing and either burned or hauled from the site. About half of the soil surface in the watershed remained undisturbed, 30% suffered moderate disturbance or compaction, and 20% was severely disturbed. Limb-size material was removed from the stream and piled above the high water line; no uncut buffer was retained along the stream. Clearcutting increased the concentrations of nitrate in soil solution and stream water by 1 to 2 orders of magnitude (Sollins and McCorison 1981). In the second post-cutting year, nitrate-N concentrations averaged about 0.3 mg/L (0.1 m soil depth), 0.3 mg/L (0.2 m), 0.5 mg/L (1.0 m), and 0.6 mg/L (2.0 m). Nitrate-N concentrations averaged 0.07 mg/L in stream water. These concentrations are high relative to the uncut forest but very low relative to drinking water standards.

Middle Fork of the Santiam River, Oregon

Sullivan (1985) examined changes in suspended sediment concentrations resulting from roading and harvesting of an 8000 ha portion of the watershed of the Middle Fork of the Santiam River in the Oregon Cascades. Over a 9-yr period, a large road network was constructed in the old-growth dominated watersheds and much of the area was logged. The mainline haul road (8-10 m wide) was heavily ballasted and surfaced with crushed rock; surface runoff drained directly into the river or into major tributaries. Most of the rest of the road system was constructed as all-weather secondary roads of 4-6 m width with rock surfacing and drainage design to handle a 25-yr storm. Most side roads had slopes of 5 to 12%, with some as steep as 18%. Total road length was 179 km, giving a road density of 3.0 km/km^2 (roads occupied 4.4% of the area). Logging systems used high-lead cable yarding (upslope), with tractor yarding on gently sloping sites. Harvest units were about 20 ha in size, but progressive logging of the watersheds in adjacent units led to larger, contiguous areas of disturbance. A total of about 43% of the 8,000 ha were logged; about one-third of the logged areas burned.

Sediment yield averaged about 1.6 Mg/ha annually for 9 yr for the area upstream of the study site, and about 1.3 Mg/ha for the study site (the upstream slopes were thought to be less stable than those in the study area). Most of the sediments were transported during storm events. Suspended sediment concentrations averaged (over 9 yr) about 71 mg/L for the stream above the study site, compared with 50 mg/L for the study site. Apparently the water draining the study site had lower sediment concentrations than the upstream water, indicating either deposition of sediments or dilution of sediment concentrations from the upstream area. Turbidity was similar for the upstream river and the river flowing from the study site, remaining below about 30 NTU at all times. The cumulative impacts of road building and harvesting of 43% of the 8,000 ha study area appeared negligible.

Alsea Watershed, Oregon

The Alsea Watershed area is about 12 km south of Toledo, Oregon, in the Coast Range. Slopes average 35-40%, and soil parent materials are marine sands and mudstones (Brown et al. 1973; Fredriksen et al. 1975; Fredriksen and Harr 1979). The climate is milder than at the H.J. Andrews, with a longer growing season at lower elevation (135-490 m), higher precipitation (about 2600 mm/yr), and higher streamflow (about 1900 mm/yr). The forests are dominated by mature Douglas fir and red alder. In 1966, about 80% of the 71-ha Needle

Branch watershed was clearcut. A road system built along ridges was used to avoid the deeply incised drainages. About 25% of the 303-ha Deer Creek watershed was clearcut in 3 blocks. Broadcast fires were used to reduce slash in all clearcuts. Unharvested Flynn Creek (202 ha) served as a control.

Sediment concentrations increased markedly in the clearcut Needle Branch watershed after the hot slash fires, with only a minor contribution from road drainage and erosion of sidecast soils (Beschta 1978). The control watershed showed annual sediment concentrations of less than 7 mg/L and a maximum daily concentration of 100 to 400 mg/L for 1966-1968. The Needle Branch sediment concentrations averaged 8 mg/L (1966, roads constructed); 16 mg/L (1967, clearcut and burned); and 10 mg/L (1968, post treatment). The maximum daily average reached 1260 mg/L in the first year after harvesting and burning, compared with 100 mg/L in the control watershed. In contrast with the completely harvested watershed, the 25% clearcut (with buffer strips) in Deer Creek showed only moderate increases in stream sediment yields in 2 out of the first 8 yr of water treatment (Beschta 1978).

Nitrate-N concentrations in the control watershed were relatively high, with an annual average of about 1.2 mg/L. Peak values were about 3.2 mg/L and annual nitrate-N output was 25 to 35 kg-N/ha (Brown et al. 1973). Nitrate-N concentrations in Needle Branch were substantially lower than in the control watershed before harvest (about 0.4 mg/L), but they increased by about 4-fold after clearcutting (1.5 mg/L, with peak of 2.1 mg/L and annual output of 15 kg-N/ha). By 1990, average concentrations had declined back to 0.4 mg-N/L (Stednick and Kern, 1992). Concentrations in Deer Creek were similar to the control watershed and showed no response to patch cutting.

The monthly average water temperature was increased by 8 °C following clearcutting and burning of Needle Branch, and the maximum observed difference was 16 °C (Brown and Krygier 1970).

Increases in stream temperature after forest cutting decline as revegetation proceeds. Summers (1983, cited in Beschta et al. 1987) evaluated the rate of increase in canopy coverage for major vegetation types in western Oregon. Coast Range conifer forests generally return to preharvest shade levels within about 10 yr, compared with about 20 yr for Douglas fir forests in the Cascade Mountains. High elevation forests of hemlock, Douglas fir, and Pacific silver fir may take longer than 30 yr to reach preharvest shade levels.

Coast Range, Oregon

Miller and Newton (1983; Miller 1974) examined the effects of clearcutting, slash burning, and herbicide application on nitrate losses from mixed forests of Douglas fir and nitrogen-fixing red alder. Precipitation in the region is very high, averaging about 2000-2700 mm/yr, and temperatures are mild (monthly means are from 4 °C in January to 20 °C in July). Slopes are up to 100%. Miller and Newton (1983) examined water chemistry in a total of 14 streams in 3 locations in the Coast Range. At Siletz Creek, two streams were sampled as controls, and two watersheds were (1) sprayed with a mixture of 2,4-D and 2,4,5-T to kill alder; (2) harvested to remove Douglas fir logs; and (3) planted with Douglas fir seedlings. At Drift Creek, two streams served as controls. On two watersheds there, both alder and Douglas fir were cut; the Douglas-fir logs were removed; residual vegetation was sprayed with herbicides; and the site was broadcast burned. At Brush Creek, two streams were retained as controls and two watersheds received a total of three applications of herbicides to kill alder and understory shrubs with no cutting of Douglas fir. Another Brush Creek watershed was clearcut harvested, and herbicides were sprayed on about 60% of the watershed. The fourth Brush Creek watershed was sprayed with herbicides, clearcut, and had Douglas fir logs removed.

Nitrate-N concentrations at Siletz Creek showed no response to treatments, averaging 0.6 mg/L for all 4 creeks (calculated from Miller 1974, table 2). The combination of moderate nitrate concentrations and high runoff combined to give relatively high rates of nitrate loss, from 12 to 17 kg of N/ha annually.

Nitrate-N concentrations at Drift Creek were higher than in Siletz Creek in both control and treated watersheds, with annual averages ranging from 1.5 to 2.0 mg/L with no effect of treatment evident. Maximum observed nitrate-N concentrations were about 3.0 mg/L in all watersheds, which is about 30% of the drinking water standard of 10 mg/L. Annual loss of nitrogen from nitrate in stream water ranged from 17 to 41 kg/ha.

Nitrate-N concentrations were highest at Brush Creek, averaging between 0.7 and 2.1 mg/L across all treatments. Maximum observed concentrations were about 3.5 mg/L, which is again well below drinking water standards. The combination of higher nitrate concentrations and high discharge combined to produce annual nitrogen losses of 17-74 kg/ha.

Bull Run Watersheds, Oregon

The Bull Run River basin has been the primary water source for Portland, Oregon, since the late 1800's (Rinella 1987). The 26,000 ha watershed is located about 45 km east of Portland and was declared a reserved area to protect water quality in 1892. Land management within the Reserve has varied; 17,000 ha were opened to recreation and logging in 1959 and were closed in 1976. Now the reserve is open to management within strict guidelines to protect water quality. Elevation ranges from 230 to 1400 m. At lower elevations, precipitation between October and April accounts for about 80% of the annual total of 2040 mm/yr. Interception of fog by tree canopies may add about another 880 mm/yr to these watersheds (Harr 1983). Total precipitation at upper elevations is probably double that of the lower elevations. Most of the basin consists of volcanic rock with some minor sedimentary deposits; slopes are steep, especially near the deeply incised streams.

Virtually all of the major subbasins in the Bull Run Watershed have been logged to some extent. High-lead and skyline yarding systems have been used exclusively to remove logs from both patch cuts and shelterwood cuts. Slash has been broadcast burned or piled and burned.

Water quality around the watershed is very good. Suspended sediment averaged (1978-1983 in the Main Stem) 0.4 mg/L, turbidity 0.2 NTU, and nitrate-N 0.17 mg/L. In an extensive examination of patterns in water quality in the watershed, the only water chemistry feature that related significantly to proportion of the area cut in the previous year was specific conductance, which increased slightly with increasing area harvested (Rinella 1987).

Two Fox Creek watersheds (59 and 71 ha) were partially clearcut (25% of the area), with broadcast burning and without (Fredriksen et al. 1975; Harr and Fredriksen 1988). From 1969 through 1973, the suspended sediment concentrations in the control creek ranged from 0.6 to 2.4 mg/L, with a maximum 24-hour average of 1.4 - 7.3 mg/L. Sediment concentrations were not increased by the harvesting or burning treatments. Nitrate-N concentrations for the control stream averaged between about 0.003 and 0.006 mg/L from 1970 through 1981, with peak concentrations up to 0.06 mg/L. The nitrate-N concentrations for the 25% cut and burned watershed reached peak concentrations of 0.08 mg/L and averaged less than 0.012 mg/L after about 2 yr. The logged but unburned watershed showed the highest nitrate-N concentrations, with a peak of up to 0.3 mg/L and sustained high yearly averages of about 0.04 mg/L.

Coyote Creek, Oregon

Four watersheds were studied in the Coyote Creek Experimental Forest in southwestern Oregon to determine the impacts of a variety of harvest methods on water quality (Harr et al. 1979; Adams and Stack 1989). The watersheds range from 49 to 69 ha, at elevations between 730 and 1065 m. Parent materials are deep deposits of welded and non-welded tuffs, with andesite and basalt bedrock on ridges. Side slopes range from 20 to 80%. Soils include Typic Haploxerults, Distric Xerochrepts, and Dystric Haploxerolls and Haploxeralfs. Soil depths range from about 0.6 to 1.8 m. Precipitation averages 1170 mm/yr, with 80-90% falling between October and March. The 100- to 300-yr-old mixed conifer forests were dominated by Douglas fir, ponderosa pine, sugar pine, incense cedar, western hemlock, grand fir, and big leaf maple.

Three harvest systems were applied to separate watersheds: shelterwood (50% of basal area removed); a series of patchcuts (20 patches from 0.7 to 1.4 ha, comprising 30% of the watershed area); and a complete clearcut. All logs were yarded by tractors in the shelterwood cut watershed. About half of the patch cuts were yarded by tractor and half with a mobile, high-lead cable system. Most of the clearcut watershed was yarded with a high-lead cable system, but tractors were used on the lower portions of the watershed. Some slash was piled in the patch cuts, and all slash was burned. Slash was windrowed by tractor in the lower portion of the clearcut watershed and burned. The clearcut watershed was planted with Douglas fir seedlings, as were portions of the other harvest units. Natural regeneration was high, and stocking was adequate on all areas within 4 yr after harvest.

Suspended sediment concentrations averaged less than 40 mg/L for the control, shelterwood, and patch-cut watersheds, with no increases resulting from harvesting activities. Average sediment concentrations in the clearcut watershed increased to 170 mg/L (1450 mg/L maximum 3-week average) in the first year after logging, and to 270 mg/L in the third year (2,300 mg/L maximum 3-week average). No increases were evident in later years. Nitrate-N concentrations were lowest in the shelterwood watershed, averaging < 0.015 mg/L. The control and patch-cut watersheds averaged between 0.015 and 0.04 mg/L. Nitrate-N concentrations were increased substantially by clearcut, reaching maximum annual levels of 0.1 mg/L (maximum 3-week average 0.5 mg/L). Stream water temperatures were increased only in the clearcut watershed, where summer maximum temperatures were about 8 °C higher than other watersheds in the first year after logging. Temperature differences declined slowly each year, with about a 3 °C difference 8 yr after harvesting. Overall, the shelterwood and patch-cut treatments had no substantial effects on water quality, and the impacts of clearcutting were generally minor with the exception of high sediment loads during peak flow periods.

High Ridge Watersheds, Oregon

The High Ridge Watersheds are representative of much of the upper slope conifer forests of eastern Washington and Oregon (Tiedemann et al. 1988). The harvesting studies involved 4 small, adjacent watersheds that range in size from 24 to 118 ha about 22 km northwest of Elgin, Oregon. Elevations range from 1440 to 1620 m, with slopes of 2-25%. The area is underlain by basalt, but the soils (Typic Vitrandepts and Cryandepts) have formed primarily in recent (6700 yr old) volcanic ash that ranges in depth from about 0.7 to 2.5 m. Precipitation averages about 1400 mm/yr, concentrated as snow during the non-growing season; runoff averages about 470 mm/yr. Forests are densely stocked with grand fir, Engelmann spruce, subalpine fir, western larch, Douglas fir, and lodgepole pine.

Three of the watersheds were harvested in 1976, with the fourth retained as a control. About 40% of Watershed 1 was clearcut in 2 blocks of 3.6 and 8.5 ha, and residues were piled by tractors and burned in September 1976. Unburned slash was then spread across the watershed by tractors. Half of the clearcut was seeded with grasses, and the other half was left unseeded. Watershed 2 received a selection cut that removed about half of the stand volume; residues were yarded to a landing below the watershed and burned, and no seeds were spread. Watershed 4 received 10 patch cuts (0.8 to 2.4 ha), which covered about 17% of the watershed. Slash was piled and burned as in Watershed 1, and half of the patch cuts received grass seeds.

Annual average nitrate-N concentrations were less than 0.015 mg/L in the control watershed and between 0.03 and 0.1 mg/L in the harvested watersheds. The maximum observed nitrate concentration was relatively low—0.3 mg/L in Watershed 1 (large clearcut blocks).

Hansel Creek, Washington

Hansel Creek has been the location of intensive studies on the effects of forest practices on water quality, with Andrews Creek serving as a control (Fowler et al. 1988). The watersheds are located about 30 km west of Wenatchee, Washington, at elevations ranging from 560 to 2130 m. Precipitation averaged from 830 to 1530 mm/yr from low to high elevation stations. A snowpack of about 2 m is common, and some summer months receive no precipitation in some years. The watersheds sit atop a variety of rock types, including sandstones, ultrabasic serpentine, schist, and granite. Gravel till covers much of the parent material, with soil depths ranging up to 1.5 m. On south- and west-facing slopes, the old forests were dominated by ponderosa pine, Douglas fir, western larch, and western white pine. On east-facing slopes, grand fir and Engelmann spruce were more dominant. Erosion hazard was expected to be high, so logging operations utilized longspan skyline yarding systems, with some helicopter yarding. Four watersheds, ranging from 34 to 170 ha, were partially harvested (from 22 to 47% of the area) with a series of clearcuts.

Prior to road construction, suspended sediment loads averaged about 4 mg/L, with turbidity of about 1 NTU. After road construction, sediment concentrations averaged about 180 mg/L (and 24 NTU) immediately below the road, with peak concentrations of 500 mg/L (and 70 NTU). Sediment loads returned to control levels within about 2 yr. Nitrate-N concentrations averaged about 0.15 mg/L in control watersheds and about 0.04 mg/L in harvested watersheds (no significant difference). Two streams in partially harvested watersheds showed temperature increases of about 1 ^{0}C in summer, and two showed no change. Fowler et al. (1988) concluded that the application of best management practices to logging in the eastern Cascades should adequately protect water resources.

Thompson Research Center, Washington

The Thompson Research Center, about 50 km west of Seattle, has been the site of intensive nutrient cycling research by D. Cole and colleagues. Mean annual temperature is about 10^{0} C, with precipitation of 1300 cm that falls mostly as rain in fall and winter (Van Miegroet et al. 1990). Comparisons of soil solution chemistry from beneath stands of red alder revealed extremely high concentrations of nitrate-N, averaging from 5 to 17 mg/L annually (Bigger and Cole 1983; Johnson and Lindberg 1992). Concentrations of nitrate under Douglas fir stands were extremely low (< 0.015 mg/L annual average). In several alder-harvesting studies, nitrate concentrations dramatically decreased after clearcutting (Bigger and Cole 1983; Van Miegroet et al. 1990), to less than 0.015 mg/L within 4 yr after harvest. These studies have not been duplicated elsewhere in the region, but the herbicide treatments examined by Bigley and Kimmins (1983) and Miller and Newton (1983) found either no change or increases in nitrate concentrations after alder suppression. More work is needed to determine the patterns of nitrate concentrations in alder stands (across ages, sites, and harvesting regimes) and the mechanisms responsible for the patterns.

Clearwater River, Washington

Declining coho salmon populations in the early 1970's led to the establishment of a major research focus on impacts of forest practices on fishery habitat in the Clearwater River basin (375 km^2) on the Olympic Peninsula (Cederholm and Reid 1987). The steep, forested slopes are underlain by siltstones and sandstones. Precipitation averages about 3500 mm/yr and is concentrated between November and March. Since 1975, road construction has been designed to minimize erosion and sidecast (road cut material piled on the downhill side of the road). Uncut buffer strips (10-100 m wide) have been retained along larger (third order or greater) streams. About 60% of the basin has been logged at least once. Declines in coho salmon populations in the 1950's and 1960's were attributed to increased harvesting in the ocean, but reduced fishing pressure did not return populations to target levels. The major impacts of forest practices in the basin were associated with stream sedimentation. High concentrations of fine particles (< 0.85 mm) reduce the suitability of gravel beds for spawning sites. In the tributaries of the Clearwater River, fine particles comprise from 6 to 23% of the stream bottom materials, compared with 3 to 10% for streams draining undisturbed watersheds. The higher fine concentrations occurred in the more heavily logged and roaded watersheds (fig. 21). The particle size distribution in stream bottoms is not a direct measure of water quality, but it does represent a cumulative effect of changes in water quality with strong implications for fish habitat. The successful survival of coho salmon eggs through to emergence of fry declined dramatically from about 65% with 5% fine particles to less than 20% with 15% fine particles. Suspended sediment also reduced the ability of juvenile salmon to catch insects; concentrations of just 100 mg/L reduced insect catch by 40%.

The major sources of stream sediments were landslides and erosion from road surfaces (table A.7). The basin-wide average yield of 250 Mg/km² is about 3 times the expected value for undisturbed basins.

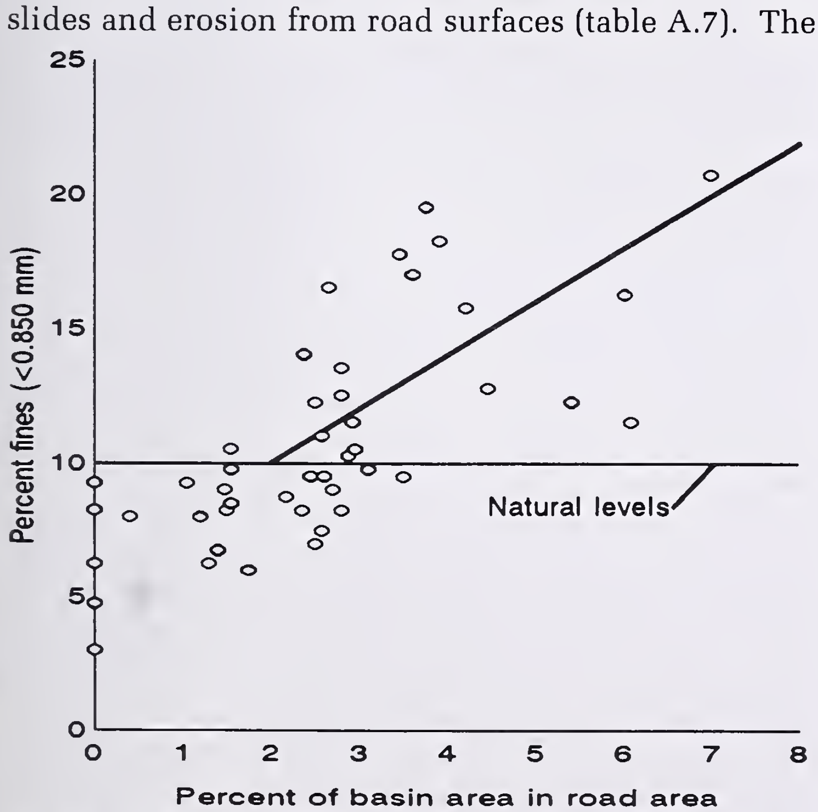


Figure 21. Relation between percent of fines in gravel beds and the percent of the basin in roads for Clearwater River watershed ($r^2 = 0.62$) (from Cederholm and Reid 1987).

UBC Research Forest, British Columbia

Two watersheds were examined by Feller and Kimmins (1984) in the University of British Columbia's Research Forest near Haney (about 60 km east of Vancouver). The elevations of the watersheds range between 140 and 450 m above sea level, with slopes of about 10-20%. The soils are shallow, coarse-textured Typic Haplorthods. Precipitation averages about 2150 mm/yr, with a mild annual average temperature of 9 °C. The pre-harvest forests were dominated by 70- to 90-year-old western hemlock, western red cedar, and Douglas fir, but young (15 yr old) Douglas fir plantations covered about one-third of each watershed. The forests were commercially clearcut in 1973, using both tractors and high-lead cable yarding systems. About 60% of the 23-ha Watershed A was clearcut, compared with 19% of the 68-ha Watershed B (immediately above the weir). Road construction was minimal (about 350 m in Watershed A and 200 m in Watershed B), and logging activities avoided impacts on the stream channels (some logs were yarded across the streams). Slash was broadcast burned in Watershed B, and both areas were planted with Douglas fir seedlings in 1975.

Annual average nitrate-N concentrations in the control watershed ranged from 0.015 to 0.07 mg/L between 1972 and 1982. The average for the clearcut Watershed A jumped to 0.5 mg/L in the first post-harvest year, compared with just 0.17 mg/L for the same period in the harvested and burned Watershed B. By the third post-harvest year, nitrate concentrations were as low or lower than the control watershed for both harvested watersheds.

Maximum observed temperatures for the control stream remained less than 17 °C, but the stream in the clearcut watershed had temperatures in excess of 17 °C for about 5 days in the first summer after cutting (Feller 1981). The increased temperatures in the clearcut and burned watershed were greater and lasted longer than those in the clearcut watershed.

In 1982, a helicopter application of glyphosate to a 10.5 ha portion of the unburned watershed killed about 80% of the red alder and most of the salmonberry understory (Bigley and Kimmins 1983). After 4 months, soil solution concentrations of nitrate-N increased about 3-fold, from about 0.7 mg/L in the control areas to 2.1 mg/L on the herbicide-treated portion of the watershed.

Okanagan Valley, British Columbia

Hetherington (1976, cited in USEPA 1980) examined stream water chemistry above and below a 960 ha area that contained several clearcuts totaling 155 ha. The maximum observed nitrate-N concentration was 0.4 mg/L below a clearcut, with a mean concentration of 0.03 mg/L. A control watershed had a maximum nitrate-N concentration of 0.12 mg/L, and a mean of 0.03 mg/L.

Carnation Creek, Vancouver Island, British Columbia

Carnation Creek drains a watershed of about 1000 ha on the west coast of Vancouver Island (Hartman et al. 1987). A wide valley bottom is bordered by slopes of 40-80%. Soils on the slope are shallow (< 0.7 m) gravelly loams, and those of the valley bottom are deep alluvial sands and gravels. Precipitation averages about 3500 mm/yr. The stream and fish populations have been studied continuously since 1971, including the period between 1976 and 1981 when about 40% of the watershed was logged. Dissolved ion concentrations increased after logging and slashburning but declined within 3 yr. Logging had substantial effects on the structure of the stream banks and bed. In a portion where a buffer strip was retained along the stream, the width of the stream increased by about 0.6 m in 7 yr; the rate of bank loss was about 93 m^2/km of stream length annually (Powell 1988), or a total of about 0.5 m^3 of material lost for every meter of stream length (Scrivener 1988). In a portion where no buffer strip was retained but logs were yarded carefully to minimize stream damage, the width of the stream increased by more than 8 m in 6 yr, and about 1650 m^2 of stream bank was lost per each kilometer of stream length annually. More recent observations indicate that as a result of some large storm events, changes are even greater than these published results (S. Chatwin, B.C. Ministry of Forests, personal communication). The amount of fine particles (clay, silt, and fine sand) increased from about 5% of the material comprising the streambed to about 9%. Water temperatures increased substantially after logging (fig. 22) throughout the year (Holtby 1988a,b). Higher water temperatures in the late winter and early spring allowed earlier fry emergence and smolt migration, perhaps doubling the biomass of coho salmon smolts for several years after logging (assuming early migration does not increase mortality in the ocean).

Water Quality Impacts of Forestry Activities in Alaska

Gibbons et al. (1987) summarized the history of studies focusing on the interactions of forest practices and fisheries in southeastern Alaska. In the 1950's and 1960's, studies focused on the effects of logging on stream bank stability at a few intensively studied sites, with a strong emphasis on salmon species that have only

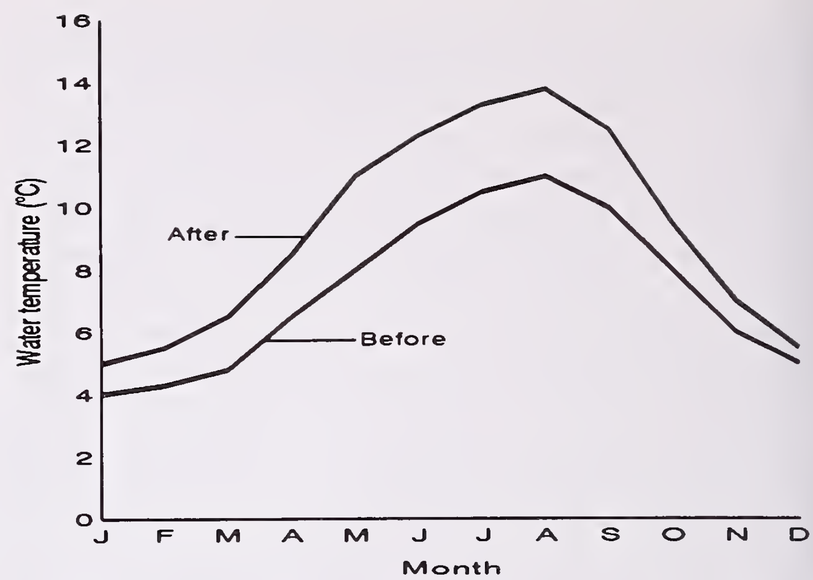


Figure 22. Mean monthly temperatures for stream water before and after clearcutting at Carnation Creek (from Holtby 1988 a, b).

short residence time in streams before migration. More recent work has broadened the scope to include long-residence species and many more locations. The findings of this early work concluded that productivity of spawning beds depended strongly on the permeability of the gravel beds, which in turn depended on the percent of fine particles. Permeability declined 10-fold as fine particles increased from about 6% to 12% of the gravel bed. Logging practices increased the fine particle content of gravel beds, but the effect lasted less than 5 yr and the connection with lower survival rate for salmon eggs and fry was unclear. Logging increased monthly mean temperatures by about 2 °C (Meehan 1970), with maximum observed temperature increases of about 5 °C.

Meehan et al. (1969) intensively studied a series of watersheds in the Maybeso Experimental Forest (Tongass National Forest) near Hollis on Prince of Wales Island (75 km west of Ketchikan). Temperatures average near freezing in winter and 16 °C in summer; precipitation averages about 2600 mm/yr. The glaciated valleys have steep slopes (from sea level up to about 1000 m) with unstable soils (0.3 to 1 m deep) on top of sedimentary (shale black argillite, graywacke, conglomerate) parent materials. The forests are dominated by western hemlock and Sitka spruce, with some Alaska yellow cedar and western redcedar. About 20% of the Harris River watershed and 25% of the Maybeso watershed were logged. Logs were removed with a high-lead yarding system, with some tractor skidding on the flat valley bottoms. Roads were constructed carefully: away from streams, with gravel beds and surfaces, and with log bridges for stream crossing. Stream temperatures increased by about 0.5 to 2 °C, depending on season and which stream was used as a control. Logging had little effect on sediment concentrations in the streams, even though debris flows were numerous in the clearcuts (more than 100 in the Maybeso valley since logging began in 1953). Only one of the debris flows reached the stream. Effects on fish populations were not clear; escapement of salmon increased after logging, but concurrent reductions in fish trapping in nearby waters confounded any response to logging.

Studies during the mid-1960's continued to document increased fine particle contents of gravel beds in streams draining logged slopes, but the large number of factors that control survival of eggs and fry prevented any clear effect of logging on salmon populations to be discerned. This prompted the Alaska Department of Game and Fish to give up monitoring impacts of logging and to begin participating in multidisciplinary projects aimed at minimizing impacts through land use planning. Stednick (1980) noted that high variations in natural processes that cause changes in sediment concentrations and turbidity obscure effects that may result from forest practices. He also suggested that enforcement actions be focused on implementation of best management practices rather than on monitoring of actual effects on water quality. In 1982, a working group was formed through Cooperative Forestry-Fisheries Research to develop the scientific basis needed to manage streamside areas. Guidelines were prepared on how land managers could protect and improve fish habitat. As part of this effort, Koski et al. (1984) examined the effects of logging with and without stream buffer strips. Retention of buffer strips kept stream characteristics similar to those found before logging; removing trees adjacent to streams led to increased periphyton production, lower channel stability, less canopy cover, lower pore volume in gravel beds, less woody debris, and fewer undercut banks. Increased productivity of streams without buffer strips might lead to greater fry production during summer, but reduction in winter habitat (from lower levels of debris and channel structural diversity) might reduce later survival.

Stednick et al. (1982) examined the effects of slash burning after harvest on water quality in southeastern Alaska by comparing water quality above and below the burned unit. Above the burned area, sediment concentrations ranged between 0.2 and 310 mg/L, with turbidity reaching a maximum of about 1 NTU. Below the burned area, maximum sediment concentrations reached 1290 mg/L, with maximum turbidities of 1 to 5 NTU. The authors could not tell if the increased sediment concentrations resulted from burning or from the prefire occurrence of two slumps in the stream bank. They conclude that harvesting followed by slash burning did not significantly affect soil or water resources.

Little work has focused on impacts of forest practices on water quality in the interior of Alaska, but continuing work at the Caribou-Poker Creeks research watersheds near Fairbanks will provide some information in the future (Hilgert and Slaughter 1987).

Slope Stability and Timber Harvesting

Road building and logging in the Pacific Northwest have been associated with major increases in slope failures, mass movements, and stream sedimentation. These effects depend strongly on site characteristics, such as slope, parent material, soils, and soil water content.

The H.J. Andrews Experimental Forest is comprised of two zones of very different slope stability. Above 1000 m, lava flow bedrock underlays the surface, and only two small, road-related landslides occurred since forest cutting began in 1950 (Swanson and Dyrness 1975). At lower elevations, altered volcaniclastic rocks are much less stable, and 139 slides occurred between 1950 and 1975. Slide erosion across 25 yr totaled about 60 m^3/ha of harvested land within the unstable zone, compared with just 20 m^3/ha for uncut areas. Slides within the road rights-of-way are 30 times greater than within uncut forests. Debris torrents (massive movements of water, soil, and debris down stream channels) also increased for cut areas and roads. Torrents were almost 5 times greater per hectare of harvested forest than intact forest, and over 40 times greater per hectare of roadway (Swanston and Swanson 1976).

Swanston and Swanson (1976; see also Swanston 1991) provided similar summaries for other areas within the region:

- Cut areas in Stequaleho Creek on the Olympic Peninsula in Washington showed no increase in slides, but slides along roads were 12,000 times greater than in uncut forests.
- Cut areas in the Alder Creek drainage in the Oregon Cascades had a slide rate that was 2.6 times the slide rate of uncut areas, and slide rates along roads was 350 times greater. Debris torrents were increased 9-fold in cut areas, and over 130-fold along roads.
- Cut areas across a variety of drainages in coastal British Columbia had slide rates about double those of uncut areas, and slide rates for roads were about 25 times greater.

Schroeder and Brown (1984) examined the occurrence of debris torrents following an intense storm (about 150 mm in 24 hours, with an estimated return interval of 5 to 7 yr) in the Coast Range in Oregon. In two drainages, new landslides occurred at about one for every 3.5 ha of clearcut area, compared with one slide for every 16 ha of uncut area. About 20% of all slides originated along roads.

Bourgeois (1978) documented that the majority of slides on steep terrain on Vancouver Island originated along roads. On slopes greater than 55%, the number of slides per kilometer of road varied from about 0.15 to 1.2. Landslides within clearcuts averaged about one slide for every 115 to 150 ha of land with slopes over 55%.

Fertilization Effects on Water Quality

The effects of fertilization on stream chemistry have varied substantially among studies (reviewed by Bisson et al. 1992). Fredriksen et al. (1975) tabulated the results from 6 studies. At Coyote Creek in southwest Oregon, fertilization with 225 kg urea-N/ha led to peak concentrations of urea-N (1.4 mg/L) and ammonia-N (0.04 mg/L) about 2 days after fertilization (fig. 23). Nitrate-N

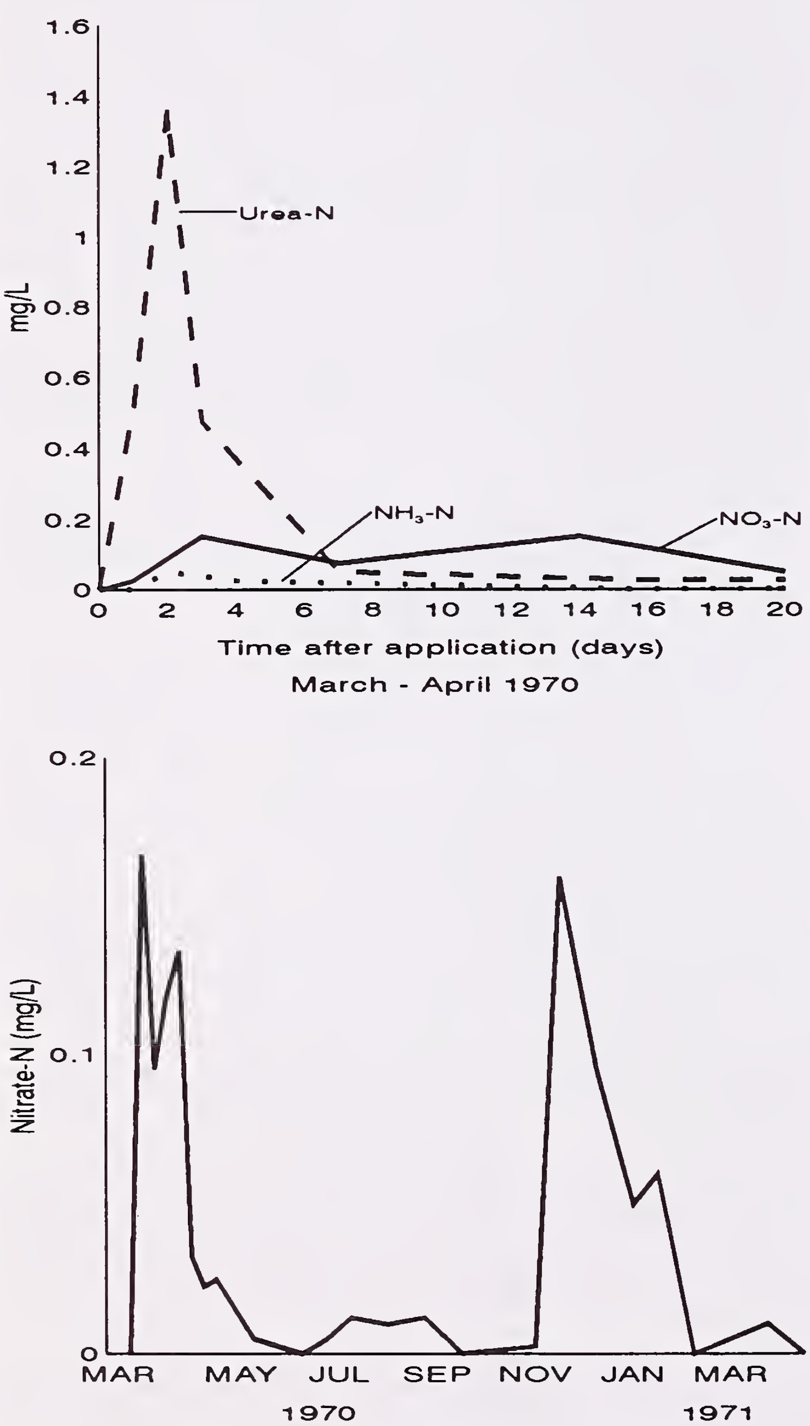


Figure 23. Response of stream water N concentrations to fertilization with 225 kg-N/ha as urea (from Fredriksen et al. 1975).

concentrations peaked after 3 days, at about 0.17 mg/L. A second, similar peak in nitrate concentration developed several months later with the onset of the wet autumn season. The patterns from the other 28 similar studies (USEPA 1980) included peak urea-N concentrations of 44 mg/L; 4.0 mg/L of nitrate-N; and 1.4 mg/L of ammonia-N (not clear in the summary if this is truly ammonia-N, or ammonium+ammonia-N; such a high value suggests that it must be ammonium+ammonia-N). Fredriksen et al. (1975) concluded that fertilization does not raise N concentrations to toxic levels or pose any threat to stream water quality.

Tiedemann et al. (1978) examined the effects of fertilization (after a wildfire) on stream chemistry in the Entiat Experimental Forest in north-central Washington, and an effect was detectable only immediately after application when nitrate-N concentrations rose to 0.15 mg/L.

Hetherington (1985) examined stream water nitrogen concentrations following a commercial fertilization operation on Vancouver Island, British Columbia; no effort was made to minimize application over streams within the unit. Nitrogen was applied as urea. Urea-N concentrations peaked at 14 mg/L 12 hours after fertilization and declined to undetectable levels within 6 days. Ammonia-N concentrations peaked at about 0.56 µg/L within 24 hours of application and declined to pretreatment levels in 13 days. Nitrate concentrations peaked about 2 months after application as heavy rains drained the soils. The maximum nitrate-N concentration reached 9.5 mg/L, very close to the 10 mg/L drinking water standard for the United States. Nitrate concentrations returned to pretreatment levels during the first winter, with a minor peak occurring 1 yr after fertilization.

Meehan et al. (1975) documented increased nitrogen concentrations in stream water following fertilization with 210 kg N/ha as urea in 2 watersheds in southeast Alaska. In the Falls Creek unit, nitrate-N concentrations increased to a peak of about 1.3 mg/L within 1 month, with peaks up to 1.7 mg/L through the first year. The Three Lakes Unit showed one spike of 2.4 mg/L about 6 weeks after fertilization but no substantial increases at any other time. Ammonia increased in Falls Creek to a maximum of < 3 µg/L, well below any toxicity thresholds.

Bisson (1982) examined stream water concentrations of N following fertilization at rates of 65 to 225 kg-N/ha as urea. Maximum observed concentrations were 37.8 mg/L of total N (mostly as urea, on the day of application); 2.7 mg/L of nitrate-N (2 months after fertilization); and 0.4 µg/L of ammonia-N (one week after fertilization). Similar studies reported by Bisson (1988) included similar maximum concentrations: 40 mg/L as urea-N, 2.0 mg/L as nitrate-N, and 0.3 µg/L as ammonia-N.

Herbicide Effects on Water Quality

A range of herbicides has been used extensively in the Pacific Northwest over the past several decades to improve regeneration and growth of conifers. Herbicide residues in streams peak within hours of application if direct application above streams occurs (Fredriksen et al. 1975). A summary report for the EPA (Newton and Norgren 1977) concluded that no cases of adverse water quality impacts have resulted from herbicide applications within registered guidelines for herbicide use in forests. Norris et al. (1991) provide an excellent summary of herbicide chemistry and toxicity, observed concentrations in streams, and expected safe concentrations. Secondary effects of herbicide application are also possible:

- Killing riparian vegetation that stabilize stream banks could lead to increased bank erosion (MacDonald et al. 1991), but we know of no case studies documenting such problems in forestry operations.
- Suppression or removal of red alder may increase (Bigley and Kimmins 1983) or decrease (Van Miegroet et al. 1990) soil solution nitrate concentrations, but too little information is available for generalization.

Summary

Critical water quality aspects of forest practices in the Pacific Northwest can be grouped in three categories (table A.7). *Nitrate concentrations* appear to be far below critical limits, with the exception of:

1) nitrogen fertilization. Most fertilization studies in the region have found that increased nitrate-N concentrations following application have remained well below the drinking standard of 10 mg/L. However, one study (Hetherington 1985) found excessive levels of nitrate following heavy rains after fertilization.

2) red alder forests. Extremely high rates of nitrate leaching have been measured for red alder slopes that drain soil water and stream water. Unlike spikes in nitrate concentrations following fertilization, these high concentrations in water-draining alder forests are chronic. Removal of the alder in some cases increased (Brown and Krygier 1970), decreased (Van Miegroet et al. 1990), or had no effect on nitrate concentrations (Miller and Newton 1983). Water quality has been examined for streams draining alder forests in only a handful of stands in the Pacific Northwest, and soil solution chemistry has been documented in only a few more. Much more work is needed to evaluate the spatial extent of high stream water nitrate concentrations and the mechanisms that control these concentrations before and after harvest.

Stream water temperatures may increase dramatically after logging if streamside vegetation is removed, and in some cases warmer water may enhance survival of salmon eggs and fry (such as Carnation Creek). In general, retention of uncut buffer strips along streams prevents substantial changes in stream temperatures and protects channel structure and other features.

Sediment production is a major concern in the region, given clear evidence of declining salmon spawning success with increasing content of fine particles in spawning beds. Impacts of forest practices have been variable, depending on factors such as slope characteristics, soils, parent material, and road construction practices. Harvest-related slope failures are important in some locations (such as some geomorphologic units at the H.J. Andrews Forest), whereas stream bank changes are more important at others (such as Carnation Creek). Management of sediment production will continue to be a major feature of forest impacts on water quality in the region.

As a final note, one of the major sets of impacts of forest practices on fisheries habitat comes from alterations of channel structure and stability. Habitat quality can be degraded by failure of stream banks and by reductions in the amount of large woody debris that contributes to diversity of streambed structure. Similarly, the impact of sediment production on fish may depend more on the accumulation of fine particles in gravel streambeds rather than directly on the presence of sediment in the water. These impacts of forestry may be particularly important, but they cannot be quantified in terms of changes in the composition of water flowing from a watershed (MacDonald et al. 1991). Water quality guidelines may need to consider these changes in physical structure of streams as a result of forest practices.

Chapter 8

Pacific Southwest

About 30% of New Mexico, Arizona, and California are forested, with rangelands comprising an additional 55% (USDA Forest Service 1982). Half of the forested land is federally owned, compared with 45% of the rangelands. The majority of forests are low in productivity, with about 60% producing less than 1.4 m^3/ha of wood annually. Only 5% of the forests produce more than 8.4 m^3/ha annually, concentrated near the coast in California. Pinyon/juniper forests are the most extensive in the region, covering about 33% of the forest land, with chaparral vegetation accounting for another 13%. Ponderosa pine forests contribute the largest share among large tree species (about 20% of the total forestland). Other contributions include about 9% for spruce/fir types, 6% for Douglas-fir, and 1% for redwoods. Rangelands are in poor condition across the region, with over 70% of the rangeland classified as being in poor or very poor condition from poor grazing management.

The geography of the Pacific Southwest is highly varied, ranging from flat, low-elevation deserts to rolling high-elevation mountain "islands." Annual precipitation ranges from less than 250 mm/yr for low elevations of southern Arizona and New Mexico to more than 2500 mm/yr in parts of northern California. Runoff shows a similarly wide range, from near 0 to over 1500 mm/yr.

Published studies dealing with forest practices and water quality from New Mexico have focused on grazing-related impacts, with virtually no watershed studies on the effects of forest harvest on streams (Callaham 1990). Studies from Arizona and California have included both grazing and timber studies.

Tesuque Watershed, New Mexico

The only small-watershed study from New Mexico that we are aware of is a characterization of nutrient budgets for undisturbed forests along an elevational gradient in the Tesuque Watershed (Gosz 1980). The watersheds are in the Sangre de Christo Mountains, between 2410 and 3740 m elevation. Vegetation ranges from pinyon/juniper at lower elevations to alpine tundra. Rates of nitrate-N losses from the watersheds ranged from about 0 to 0.4 kg/ha; if 0.4 kg/ha was carried in a runoff of 250 mm/yr, the concentration of nitrate-N would average about 0.14 μg/L.

Three Bar Watersheds, Arizona

Three watersheds, ranging in size from 19 to 33 ha, were used to examine how the vegetation from chaparral to grass would affect water yield and quality (Davis 1984; 1987a,b). The soils are very gravelly, sandy loams (skeletal, mixed, mesic Udic Ustochrepts). The coarse-grained granite parent material is fractured to a depth of 6-12 m. Average gradient in the watersheds is 30-40%, with 60% slopes at the upper portion. Precipitation averages about 650 mm/yr in the Three Bar Wildlife Area (within the Tonto National Forest in central Arizona), with both winter and summer rainy periods. All three watersheds were burned by a wildfire in 1959 and seeded with Lehmann lovegrasses. The chaparral shrubs resprouted vigorously, and ground cover was well established by 1962. One watershed (B) was converted from chaparral to grassland in 2 stages. In the first stage (1965), most shrubs were treated with herbicides (Fenuron and Picloram) on northeast-facing slopes (about 40% of the watershed). Shrubs that were considered desirable, such as ceanothus and buckthorn, were left untreated. The second stage occurred 7 yr later (1972), when a similar treatment (using karbutilate) was applied to the remaining 60% of the watershed. Grass cover was very good. Another watershed (F) was treated in a single step in 1969 (with an aerial application of karbutilate); grass cover was sparse (because of the well developed shrub cover) and remained sparse for 3 years after treatment because of residual soil effects of the herbicide.

Nitrate-N concentrations in stream water draining the control watershed (D) remained less than 0.3 mg/L throughout the study, typically < 0.15 mg/L. The vegetation conversion treatments produced high concentrations of nitrate that persisted throughout the study (fig. 24). Monitoring began in Watershed B about 5 yr after the first treatment was applied, and nitrate-N concentrations averaged 2.7 mg/L, reaching peaks of 9.5 mg/L. Over the 10 yr following the second stage treatment, annual average nitrate-N concentrations ranged between 3.5 mg/L and 11.9 mg/L. A peak of 18.8 mg/L was reached in the first year after the second stage of treatment. Nitrate-N losses averaged about 10 kg/ha each year for 10 years after Stage 2. Nitrate concentrations remained elevated for at least 18 yr after the first stage treatment.

The single-treatment watershed (F) also showed drastic nitrate-N increases, reaching an annual average

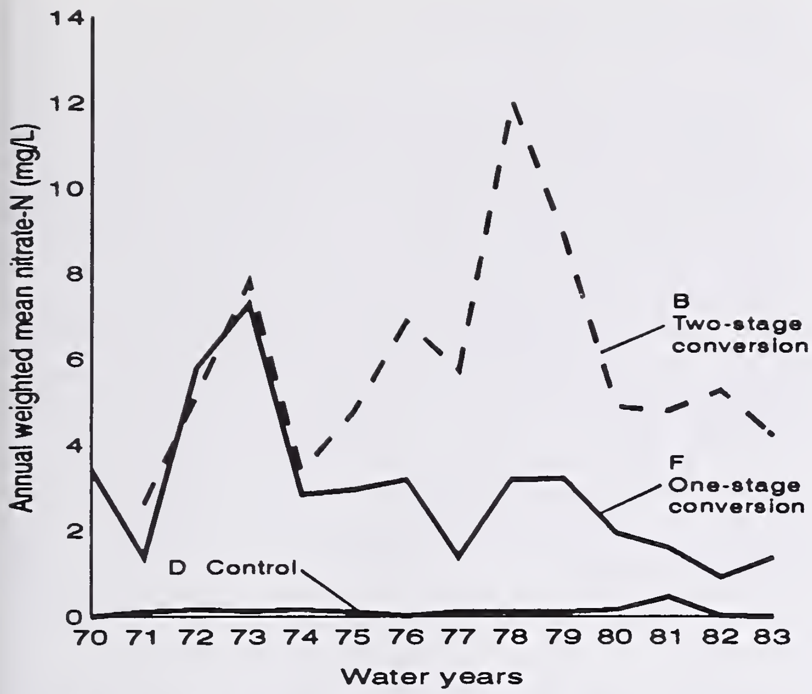


Figure 24. Nitrate-N concentrations in streams draining a chaparral watershed (D) and two watersheds converted to grassland (B,F) at the Three Bar Experimental Watersheds (from Davis 1987a). Treatments occurred in 1965 and 1972 on watershed B and in 1970 on watershed F.

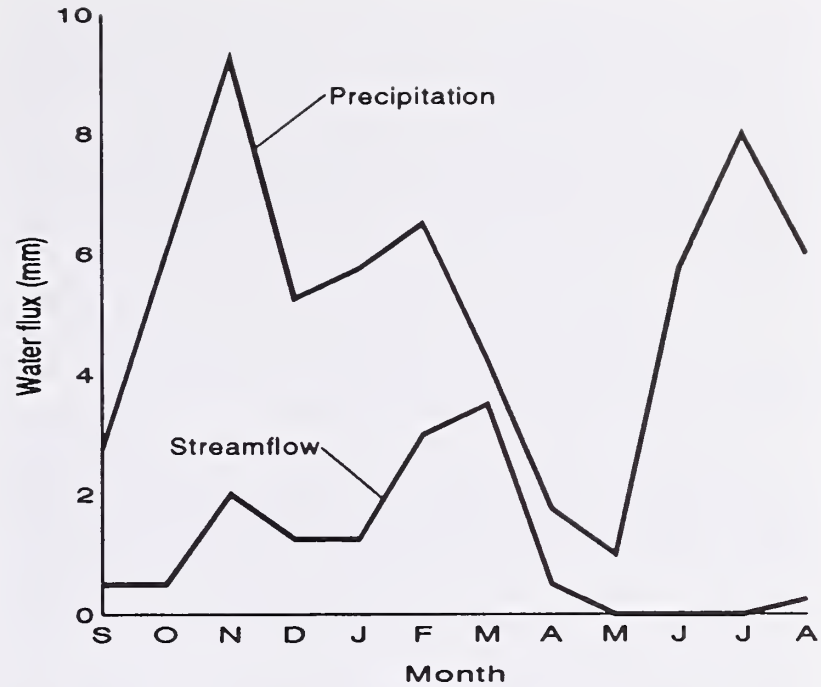


Figure 25. Average monthly precipitation and streamflow for the Beaver Creek Experimental Watersheds (from data of Brown et al. 1974).

of 7.4 mg/L 4 yr after treatment. Nitrate concentrations remained much higher than control values for the duration of the study for at least 14 yr after treatment. In 1982, the watershed was broadcast burned to control remaining shrubs. The effect of fire on nitrate losses was not clear, given relatively large annual variations over the previous 10 yr (Davis 1989).

Beaver Creek Watersheds, Arizona

A series of small-watershed experiments were developed in the 1960's at the Beaver Creek experimental watersheds in the Coconino National Forest, designed to quantify increases in water yields following forest cutting (Brown et al. 1974). Precipitation averages about 635 mm/yr, with relatively wet periods in winter and summer (fig. 25). Mean July temperature is about 19 °C, compared with -2 °C in January. The watersheds range from about 2070 to 2450 m in elevation and are underlain mostly by basalt flows (with some sedimentary limestone and sandstone exposures). Six treatments were applied in ponderosa pine vegetation, and three in pinyon/juniper vegetation. The research mandate broadened in 1971 to examine multiple resource production across the vegetation treatments. The treatments included: controls; one-third of the forest removed in strip cuts to increase snow capture; one-third of the forest removed in irregular strips; and complete tree removal and conversion to grassland.

Total dissolved solids (a measure of all ions in solution) appeared about 10% higher for the treated watersheds, but the difference was not considered significant. For the control watersheds, nitrate-N concentrations averaged about 10 µg/L over an 8-yr period. The clearcut watershed averaged 220 µg/L (M. Ryan, USDA Forest Service, personal communication), compared with 50 µg/L for the heavily thinned watershed and 20 µg/L or less for the other treatments. The grassland conversion watershed (#11) showed suspended sediment concentrations of 175 mg/L before grazing began (but after conversion), compared with 110 mg/L after grazing began (at a stocking level to consume 60% of the annual forage production).

Ward and Baker (1984) reported sediment yields from the pine harvesting experiments. Unharvested watersheds generally lost about 0.02 to 0.2 Mg/ha of sediments, but a maximum rate of 24 Mg/ha was observed for 1 yr in an untreated watershed. After harvesting of 30 to 100% of the stand basal area, sediment yields increased with increasing harvest intensity, from maximum values of 2.9 Mg/ha for 1 yr in a 31% harvest stand to 60.9 Mg/ha for the first year following clearcutting. Sediment concentrations would have been fairly low for most watersheds in most years, but extremely high sediment yields in both untreated and harvested watersheds would greatly exceed any water quality standard for suspended sediment (several thousand mg/L).

Small Scale Sediment Production in Arizona

A series of studies by Heede (1983, 1984a,b, 1987; Heede and King 1990) examined sediment production from harvesting activities on forests of ponderosa pine (at Beaver Creek) and mixed conifers in Arizona. Water quality was not assessed. Movement of sediments within subdrainages was slight, reaching maximum rates of about 0.3 Mg/ha annually from the most severely disturbed sites. Heede and King (1990) noted accumulations of sediments in the stilling pond behind a weir at the Thomas Creek site in the White Mountains of Arizona, indicating some substantive impacts on a stream channel.

Castle Creek Watershed, Arizona

Gottfried and DeBano (1990) examined the effects of prescribed fire on stream water nitrate concentrations during snowmelt in a ponderosa pine forest in eastern Arizona. Nitrate-N concentrations showed only slight responses to the burning treatment, remaining below 2.8 µg/L in all samples.

San Dimas Experimental Forest, California

Nitrate-N losses were examined from 4 watersheds (16 to 300 ha, elevations 580 to 1080 m) in the San Dimas Experimental Forest within the Angeles National Forest (Riggan et al. 1985). Slopes average about 70%, and soils are primarily shallow Typic Xerorthents developed on fractured igneous and metamorphic bedrock. Following a wildfire in 1960, two watersheds were seeded with grasses and shrub regeneration was minimized by use of herbicides. The other two watersheds contain typical chaparral vegetation that regenerated after the 1960 fire. About 20 yr after the fire, stream water nitrate-N concentrations peaked in excess of 2.8 mg/L for both grassland and chaparral watersheds. Annual nitrate-N export from the grassland watersheds were generally double those of the chaparral watersheds (7.5 vs. 3.4 kg-N/ha annual export), consistent with the pattern at the 3-Bar watersheds in Arizona (Davis 1987).

Riggan et al. (1985) also sampled over 40 streams in the San Gabriel Mountains, the Santa Monica Mountains, the Santa Ana Mountains, and the Palomar-Black Mountains following a winter storm (about 240 mm of rainfall). Watersheds along the front range that received maximum air pollution inputs showed nitrate-N concentrations of about 3.6 mg/L, compared with 1.3 mg/L for watersheds at higher elevations or farther from pollution sources. The downstream San Gabriel Valley aquifer had peak nitrate-N concentrations of 9.8 to 19.6 mg/L, and Riggan et al. (1985) concluded that increases in nitrate-N concentrations from the watersheds could play a role in increasing concentrations in the aquifer. Management of chaparral watersheds in southern California may have the potential to increase stream water concentrations of nitrate-N to the vicinity of water quality standards.

Caspar Creek, California

The Caspar Creek Watershed Study (about 10 km southeast of Ft. Bragg) is a joint project between the USDA Forest Service and the California Department of Forestry (Rice et al. 1979). Precipitation averages about 1000 mm/yr; summers are mild with frequent fogs and little or no precipitation. The 80-year-old forest of redwood, Douglas fir, grand fir, and western hemlock regenerated naturally after clearcutting and burning in the late 1800's. Two forks of Caspar Creek were calibrated against each other from 1963 through 1967, and then a road network was constructed in the South Fork. About two-thirds of the stand volume was removed through selective logging (with tractor skidding) between 1971 and 1973. The South Fork watershed is about 425 ha, underlain by sedimentary rocks. Krammes and Burns (1973) summarized the effects of road building. High sediment concentrations occurred with major storm events, ranging up to 4000 mg/L; total sediment production was increased about 4-fold after road construction. Sediment production from logging was estimated to equal about 1.4 m^3/ha. The lowest level of dissolved oxygen recorded was about 5 mg/L, about 20% lower than the equilibrium concentration with the atmosphere.

Beginning in 1985, the South Fork has been used as a control watershed, and the North Fork received a variety of treatments in 1989 (Anonymous 1987).

Redwood Creek Basin, California

Detailed watershed investigations have been carried out in the 700 km^2 Redwood Creek Basin since the early 1970's as a cooperative effort between the U.S. Geological Survey and the National Park Service (Weaver et al. 1987). Erosion and sedimentation have been the primary focus, including cumulative effects of activities on both public and private lands. Impacts have been documented within harvest units, in areas downslope of harvest units and roads, and in far-removed stream channels where sediment deposition has occurred.

Between 1954 and 1980, the total production of sediments in streams for the lower Redwood Creek Basin was estimated to be about 3.1 million m^3, derived from gullies (37%), eroded stream crossings (7%), surface erosion (4%), and streamside landslides (52%). Logging plays a major role in sediment production, particularly through increases in the average size of streamside landslides. For example, the number and frequency of streamside landslides in 16 tributaries of Redwood Creek were similar for logged and unlogged areas, but those from logged areas averaged about 4 times larger than those from unlogged areas (Pitlick

1982). More dramatically, all 44 debris slides in the lower basin during the winter of 1982-83 were directly associated with logging roads and skid trails. A major problem with road construction is the potential for rain to exceed drainage capacity of road culverts during heavy storms; in fact, culvert capacity had been exceeded at least once for about 60% of all culverts. Erosion potential of roads includes a range of historic and cumulative effects. For example, roads constructed in the 1950's and 1960's tended to have culverts that could not even accommodate 25-yr storm events, and abandonment of roads after logging has led to widespread plugging of culverts and diversion of streams over the roads. Within Redwood National Park, major efforts have been devoted to excavating inadequate culverts and retiring the roads in a sound manner.

Summary

Very little work has focused on the effects of forest practices on water quality in the semiarid Southwest (table A.8). In a review, Tecle (1991) concluded that knowledge of water quality in Southwestern forests is inadequate. He suggested this lack of information has resulted from expectations that water quality is not a forest-related problem in the region, which led to lack of investment in monitoring or research. These expectations may derive from the generally level terrain of many forest sites in the region, and very low rates of runoff. However, forest harvesting can increase erosion rates if forest floor disturbance is substantial (Heede 1987), although plot-level impacts do not represent impacts that will reach streams. The Beaver Creek study in Arizona (Brown et al. 1974) documented that high intensity storms can generate substantial sediment in streams from both forested and clearcut sites. Another substantial water quality problem identified from the 3-Bar Watershed in Arizona was nitrate increases from conversion of chaparral to grassland (Davis 1987a). Grazing may have substantial impacts on water quality in the Southwest, based on results from plot-level studies that document decreased infiltration rates and increased erosion in grazed pastures. However, direct impacts of grazing on water quality have not been examined in the Southwest.

In California, most research and monitoring emphasis has been on a scale larger than small watershed experimentation; only Caspar Creek studies have focused at the small watershed scale. Based on more extensive studies, sediment production appears to be the most critical impact of forest practices on water quality in California. The development and implementation of best management practices have included retention of buffer strips that moderate effects of harvest on stream temperatures. Remarkably little information is available on the impacts of forest practices on nitrate concentrations in stream water in California.

Synthesis

The quality of water draining forested watersheds is typically the best in the nation, whether the forests are left untouched or managed intensively. Forest practices can generally avoid significant deterioration of water quality if best managment practices (BMP's) are developed and used (see Chapter 10). In most cases, forest practices lead to minimal impacts on water quality and do not impair fish habitat or water supplies. Despite this general pattern, examples of poor implementation of forest practices are common, and degradation of water quality has occurred.

Suspended Sediment

Increased sediment loads in streams is the most widespread water pollution problem in forests, with problems reported in all regions. Sediment concentrations may drastically exceed water quality objectives even from undisturbed watersheds during rare, intense storms. The ecological impacts of increased sediment production from forest practices has received the greatest attention in the Pacific Northwest, where steep slopes, erodible soils, and valuable fisheries combine to underscore the importance of sediment production. Impacts of forest practices on sediment yields have been as great in other regions, but information on any ecological impact on fish habitat is lacking.

Roads are a major contributor to sediment concentrations in streams, and road design and maintenance are critical to minimizing of sediment problems. BMP's have been developed around the country to optimize the design and installation of road systems. Compliance with BMP's is variable among states and regions (Irland 1985; Curry 1987), and most current water quality problems associated with forest practices probably result from poor implementations of BMP's.

The most important ecological impacts of forest practices on sediment-related features involve physical changes in stream structure (MacDonald et al. 1991). These changes include increased content of fine particles in gravel beds, erosion of stream banks, increases in stream width, decreases in stream depth, and fewer deep pools. These physical features of stream structure may provide a better focus for monitoring and assessing forest practice impacts than direct monitoring of sediment concentrations in the water column (MacDonald et al. 1991).

Nitrate Concentrations

Nitrate is generally the only ion of critical interest in relation to forest practices; all other ions (such as phosphate and calcium) always remain at concentrations far below water quality standards (MacDonald et al. 1991; Salminen and Beschta 1991). BMP's might not limit nitrate concentrations; it is not clear if vegetated buffers along streams, for example, would reduce peak nitrate concentrations in all areas. But most forests show very low nitrate levels, with the following exceptions.

Streams draining red alder forests in the Pacific Northwest commonly show average nitrate-N concentrations of 1 mg/L with peaks exceeding the drinking water standard of 10 mg/L. Some alder forests show reduced nitrate outputs after harvest, and some show elevated outputs; the mechanisms behind these responses remain unclear.

The second concern involves high elevation forests of red spruce and beech in the Appalachian Mountains. Streams average about 5 mg/L at 1500 m elevation (Silsbee and Larson 1982), and soil solution concentrations average about 2-4 mg/L with higher peaks (Johnson and Lindberg 1992). Any change in these forests, such as harvesting or further decline in vigor, could elevate nitrate concentrations to the vicinity of water quality standards.

The third concern arises from chaparral and grassland watersheds in Arizona and California. Nitrate concentrations tend to be high in chaparral watersheds and increase markedly when vegetation is converted to grassland (Riggan et al. 1985; Davis 1987).

The only other concern involves northern hardwood forests; some (but not all) undisturbed forests show high nitrate concentrations in stream water, with large increases in nitrate concentrations present after harvesting.

As noted below in the "Forest Chemicals" section, forest fertilization may also lead to high nitrate concentrations.

Dissolved Oxygen

Few small-watershed studies have examined the effects of forest practices on dissolved oxygen concentrations, but these effects are relatively well understood. Oxygen concentrations in streams can be reduced both by increasing temperature and by adding readily oxidized organic matter. Very heavy inputs of fine organic debris to low-flow streams can lower dissolved oxygen levels below 1 mg/L (Brown 1989), but current forest practices generally do not add enough debris to streams to have a substantial effect (MacDonald et al. 1991).

Temperature

Removal of forest canopies over streams increases radiation inputs and can raise maximum stream temperatures by 5° C or more. Higher temperature in late winter and spring may accelerate progression among life history stages of fish and other aquatic organisms, whereas high temperatures combined with low flows in late summer could be detrimental to fish populations. Retention of buffer strips is an effective approach, though the technology of buffer strips needs further work for sufficiency, stability, longevity, and low cost (Curry 1987; Sullivan et al. 1990).

Forest Chemicals

Applying nitrogen fertilizers increases stream water nitrogen concentrations; urea and ammonia levels appear to remain well below levels of concern, whereas nitrate levels may peak at high concentrations. For Douglas fir in the Pacific Northwest, enough studies have been done to show that the risks of nitrate pollution are small (Bisson et al. 1992). Exceptions may occur (Heatherington 1985). Fewer studies are available for other regions; high stream water nitrate concentrations at the Fernow Experimental Forest indicates that more study is warranted.

Herbicide applications that follow regulatory guidelines have never been found to impair water quality (Norris et al. 1991); concentrations of herbicides in streams following forest application are generally less than 0.1 mg/L, and levels of > 2 mg/L would be needed to affect stream flora. Use of herbicides to alter riparian vegetation could have a variety of indirect effects on streams, including increased light, decreased bank stability, and altered inputs of organic matter. Little information is available on the combined indirect effects, but they are likely within the normal variations found with the development of vegetation after disturbances.

Grazing Impacts

Many studies have documented the effects of heavy grazing on riparian vegetation and soil erosion rates, but few studies have directly assessed impacts on water quality. This lack of information derives in part from the scarcity of ungrazed pastures and watersheds that could serve as controls in watershed-level grazing experiments; almost all lands in the western United States that are suitable for grazing have been intensively grazed for decades. Sediment concentrations may or may not be greater for grazed watersheds than ungrazed watersheds, but bacteriological contamination is usually greater for grazed watersheds. Management practices that protect stream banks and riparian vegetation are likely to minimize sedimentation impacts from grazing; much work remains for the development of BMP's for protecting the microbiological quality of water.

Scaling From Small Watershed Studies to Regional Scales

The direct, immediate impacts of forest practices typically occur in low-order streams (intermittent streams, and first- and second-order streams), and almost all of the information available on forest practices come from studies that focus on the scale of small watersheds (10-100 ha). This scale is important for fish habitat, but water quality issues relating to drinking water quality focus on higher-order streams. Few studies have attempted to scale-up small-watershed impacts to a regional scale, yet scaling up is critical for connecting forest impacts with drinking water quality (Binkley and MacDonald 1993). Fortunately, when BMP's are applied, most forest impacts are slight. Considerable effort would be needed to connect small-watershed level information with regional-level water quality assessments.

Nonpoint Source Pollution Control

This chapter briefly describes the laws that affect water quality of streams draining forests and rangelands and discusses the implementation of best management practices for water quality control. Much of this discussion is taken from Brown et al. (1993).

Laws Affecting Nonpoint Source Pollution from Forests and Rangelands

Federal laws influencing water quality protection on forests and rangelands can be broadly separated into two groups: general resource management laws that have some bearing on water quality, and specific water quality laws. In addition to the federal laws, many states have passed legislation resulting in nonpoint source pollution control programs affecting forest and rangeland management.

General federal resource management legislation

The Forest Service's Organic Act of 1897 states, "No national forest shall be established, except to improve and protect the forest within the boundaries, or for the purpose of securing favorable conditions of water flows, and to furnish a continuous supply of timber for the use and necessities of citizens of the United States" (16 U.S.C. 475). It is not entirely clear what "favorable conditions of water flows" was intended to mean, but conditions at the time the bill was passed suggest that it included avoiding both floods and serious erosion and sedimentation. Thus, far from being a subsidiary purpose, water quality and associated watershed protection was a key objective of National Forest reservation.

Perhaps the next highly significant law affecting national forest management was the Multiple-Use, Sustained Yield Act of 1960 (MUSYA). MUSYA codified long-standing Forest Service management policies, specifying that (subject to the constraints of the Organic Act) National Forests were to be administered for "outdoor recreation, range, timber, watershed, and wildlife and fish purposes" (16 U.S.C. 528). The MUSYA also required that production of products and services should not impair "the productivity of the land" (16 U.S.C. 531). While again emphasizing the importance of watershed protection, the act provided no specific direction on how watersheds or water quality were to be protected.

It was not until the late 1960's that general resource management legislation imposed major new environmental protection requirements on public land agencies. The National Environmental Policy Act of 1969 (NEPA) (42 U.S.C. 4321-4370) required environmental impact statements or environmental assessments for major management actions, essentially forcing consideration and disclosure of impacts, the consideration of which had previously been left to the discretion of resource professionals. NEPA required a significant increase in the documentation of potential environmental impacts. Although the required programmatic documents have often been too general to allow analysis and prediction of potential water quality violations in a particular area, the documents have helped to identify areas of concern that need to be tracked as an agency's plans become more specific.

Major public land management legislation was passed in the 1970's. The National Forest Management Act of 1976 (NFMA) (16 U.S.C. 472 et seq.) requires resource management plans for National Forests. It also contains prescriptive provisions requiring that the forest plans consider environmental effects, including water quality. The water quality provisions were largely the result of controversy over changes in water quality resulting from timber harvesting (Wilkinson and Anderson 1985). The act prohibits timber harvesting that fails to ensure that "soil, slope, or other watershed conditions will not be irreversibly damaged" (16 U.S.C. 1604(g)(3)(E)(i)). Furthermore, the agency must provide protection "for streams, stream banks, shorelines, lakes, wetlands, and other bodies of water from detrimental changes in water temperature, blockages of water courses, and deposits of sediment, where harvests are likely to seriously and adversely affect water conditions or fish habitat" (16 U.S.C. 1604(g)(3)(E)(iii)).

The Federal Land Policy and Management Act of 1976 (FLPMA) applies to all federal land, but its main effect was to provide for Bureau of Land Management land what the MUSYA and the NFMA had provided for National Forests—that is, it required planning for multiple uses while ensuring long-term sustainability. The act mentioned the "water resource" and concern for "habitat for fish" (43 U.S.C. 1701 (a)(8)) among its environmental quality provisions.

While NFMA and FLPMA placed considerable procedural restrictions on federal land management, the acts, like earlier legislation, did not impose specific constraints regarding water quality. Prohibitions against "irreversibly damaged" watershed conditions and "seriously and adversely affected" water conditions, like the MUSYA prohibition on "impairment of the productivity of the land," provide a real indication of concern and a general direction for managers. But the prohibitions by themselves lack sufficient specificity to enable clear tests of accountability regarding water quality protection. Even so, the analysis and disclosure requirements of this legislation had profound impacts on land and water management.

Specific federal water quality legislation

Water quality laws, with periodic amendments of gradually increasing specificity, and associated state laws and local ordinances, have provided the guidelines for watershed and water quality protection that the general resource management legislation lacks.

The Federal Water Pollution Control Act (PL 80-845) was originally passed in 1948, but a significant federal presence in water quality control was not initiated until the Federal Water Pollution Control Act Amendments of 1972.[3] The 1972 act (now commonly referred to as the Clear Water Act) optimistically called for the attainment of fishable and swimable waters by 1983 and the elimination of all point source discharges of pollutants into navigable waters by 1985. While the major emphasis of the act was the establishment of effluent standards for point source emissions, section 208 of the act specifically addressed nonpoint source pollution and designated silvicultural and livestock grazing activities as nonpoint sources of pollution. Section 208 required that states adopt an "areawide waste treatment management planning process" that was applicable to "all wastes generated within the area" (33 U.S.C. 1288(b)(1)(A)). The areawide plans were to include "a process to identify ... agriculturally and silviculturally related nonpoint sources of pollution, including runoff from manure disposal areas, and from land used for livestock and crop production," and to set forth "procedures and methods (including land use requirements) to control to the extent feasible such sources" (33 U.S.C. 1288(b)(2)(F)) (see Anderson 1987). The state and local plans were subject to approval by the EPA. Federal land management agencies were subject to all requirements of duly promulgated state water quality law and standards, but only to the same extent as such standards were applied to all nongovernmental entities (33 U.S.C. 1323).

Also of importance to forestry was section 404, which addressed water pollution associated with deposit of dredged and fill material. Unlike the section 208 controls, regulation of dredge and fill operations was primarily a federal function effected by the requirement to obtain a permit from the Army Corps of Engineers for discharge of dredge and fill materials into U.S. waters. The act authorized the EPA to set permit guidelines and veto individual permits (33 U.S.C. 1344). While focused on wetland protection, section 404 also regulated activities such as bridge and road construction.

Although the EPA recognized the seriousness of nonpoint source pollution early on (for example see EPA 1974, cited by Agee 1975), it initially emphasized the more serious and manageable problems of sewage treatment and industrial emissions. In 1976, some financial assistance for developing 208 plans was awarded to the states, yet implementation of section 208 plans remained a gradual process as states and localities adapted to the new goals and the developing federal-state-local working relationship.

The Clean Water Act of 1977 further amended the water quality legislation by increasing control of toxic pollutants and authorizing a program of grants to help cover the costs to rural landowners of implementing "best management practices" to control nonpoint source pollution. The 1977 amendments also exempted "normal" silvicultural activities, including road construction, from the requirement of obtaining a section 404 permit (33 U.S.C. 1344(f)), while leaving nonpoint source road construction concerns under the perview of 208 plans.

Also in 1977, the EPA formally informed states that they could elect either regulatory or nonregulatory programs for reducing nonpoint source discharges. Nonregulatory plans, adopted by most states, essentially rely on voluntary compliance and educational programs (BMP manuals, seminars, onsite inspections, etc.), sometimes enhanced by cost sharing or tax incentives. Regulatory plans impose mandatory restrictions on land management practices and allow the imposition of penalties for noncompliance. The EPA retained authority not to approve the states' areawide plans unless a state was given at least the authority to require adoption of land management practices, but the possibility of penalties became recognized as an "empty threat" (Goldfarb 1984:188).

The Water Quality Act of 1987 further amended the Clean Water Act, appropriating new funds and establishing in section 319 new requirements for states to develop and implement programs for controlling nonpoint sources of pollution (33 U.S.C. 1329). Section 208 previously required states to identify sources of nonpoint source pollution and to prepare plans to control such pollution, but it did not require that sources be related to specific bodies of water. Thus, section 208 allowed states, if they wished, to maintain only a vague link between cause and effect. The lack of specificity may have hindered plan implementation. Section 319 was intended to encourage implementation by requiring (1) detailed water quality plans that identified water bodies not meeting water quality standards; (2) identification of categories of nonpoint sources or particular nonpoint sources responsible for violation of water

[3] Other pre-1972 laws included the Federal Water Pollution Control Act of 1956 (PL 84-660) and its 1961 amendments (PL 87-88), the Water Quality Act of 1965 (PL 89-234), the Clean Water Restoration Act of 1966 (PL 89-753), and the Water Quality Improvement Act of 1970. The pre-1972 acts emphasized point sources and were essentially replaced by the 1972 amendments.

quality standards in identified water bodies; and (3) identification of BMP's to control them. Section 319 also detailed the process that the EPA was to use to either approve or disapprove the states' reports and management programs, although section 319 lacked firm criteria for determining whether a proposed management plan was acceptable. States with programs approved by the EPA could receive matching grants to facilitate implementation of the programs.[4]

Federal encouragement of water quality protection was strengthened once more with the Coastal Zone Act Reauthorization Amendments of 1990 (16 U.S.C. 1451 et seq.). These amendments to the Coastal Zone Management Act of 1972 direct the EPA and the National Oceanic and Atmospheric Administration (NOAA) to prepare "guidance for specifying management measures for sources of nonpoint pollution in coastal waters." The amendments direct the coastal states to submit a program for approval by EPA and NOAA within 30 months of publication of the guidance (16 U.S.C. 1455b).

The guidance is to include (1) a description of each "management measure" and the activities or locations for which each measure may be suitable; (2) identification of individual pollutants or categories of pollutants that may be controlled by the measures; and (3) quantitative estimates of the pollution reduction effects and costs of the measures, where "management measure" means an "economically achievable" measure for control of pollutants (16 U.S.C. 1455b).[5] The 1990 Amendments do not clarify what was meant by "economically achievable." According to the CZMA amendments, the state programs are to (1) identify coastal zone boundaries; (2) identify land uses that may cause degradation of coastal waters and management measures necessary to achieve and maintain water quality standards; (3) identify means the state will use to "exert control over" land and water uses; and (4) describe the organizational structure proposed to implement the program (16 U.S.C. 1451).

[4] In 1989, Congress appropriated $40 million for fiscal year 1990, of which $34.8 million was awarded to the states (EPA 1992, table 2). Congress appropriated $51 million for fiscal year 1991.

[5] A 126-page draft for forestry titled "Proposed Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters, Pursuant to Sec. 6217(g) of CZMA Amendments of 1990, Chapter 3: Management Measures for Forestry," was completed on April 27, 1992. The draft discusses road construction, timber harvest, site preparation, and 7 other "measures," and also lists specific "management practices" under each measure. The chapter indicates that while states are required to implement the management measures, they are not required to implement the practices, which are listed for "illustrative purposes only." States are expected to use the individual practices that best suit their specific circumstances. For example, one component of the "timber harvesting" measure is to "locate and construct landings to avoid failure of fill slopes by limiting the slope of the fill and not incorporating woody or organic materials" (p. 60). One of the listed practices for this measure says that "the slope of the landing surface should not exceed 5 percent and should be shaped to promote efficient drainage" (p. 67).

Each state program is to "provide for the implementation ... of management measures ... to protect coastal waters" (16 U.S.C. 1455b). Matching grants are available to states for developing and administering their program. Failure to submit an approvable program may lead to withholding of up to 30% of the grant funds available under both section 306 of the CZMA and section 319 of the CWA. Management of federal lands in or out of a coastal zone that affects the coastal zone waters must conform to the state program.

State nonpoint source pollution control programs

By the mid 1970's when implementation of the nonpoint source pollution provisions of the 1972 Federal Water Pollution Control Act began to take effect, some states with existing programs submitted those programs to meet the new federal requirements. Others developed new approaches, but several states, especially those with relatively few forests or with fewer perceived water quality problems on forestlands, were slow to respond.

Continuing concern about nonpoint source pollution along with the 1987 and 1990 federal legislation have encouraged more proactive state efforts at control. In the past 4 yr, additional states have adopted BMP's for forestlands and many states with programs have increased their efforts to have their BMP's understood and implemented. In addition, some states now provide cost-share funds. Others are establishing penalties for noncompliance with BMP's, especially where that noncompliance results in significant water quality degradation.

State approaches can broadly be categorized as regulatory or voluntary. Regulatory programs impose requirements on land management and allow assessment of fines and other penalties for noncompliance. States with regulatory programs tend to rely on inspection of management activities while the activities are in progress, as well as follow-up inspections, to improve compliance with BMP's and to determine whether penalties are to be assessed. Regulatory states may also require approval of harvest or road construction plans that include water quality protection measures before field work begins. States with voluntary programs emphasize education and training, including onsite inspection where requested. Increasingly, states with voluntary programs are performing formal implementation surveys to judge the success of the voluntary approach.

Four federally funded programs currently provide cost-share funds and technical assistance for forestry activities on forest or agricultural land that may have a positive effect on water quality. The Agricultural Conservation Program, begun in 1936, supports a series of agricultural conservation practices emphasizing water quality and other environmental concerns and includes such practices as tree planting, stand improvement, and animal exclusions in riparian areas. Over 7 million acres have been planted so far, mainly in the southern states. The Conservation Reserve Program, established in 1985 and expected to end in 1995, funds the retiring of highly erodible farm land through establishing permanent cover; over 2.3 million acres have been planted with trees in 41 states, with 92% of the planting occurring in the southern states. (Also, over 20 million acres have been planted in grass.) The Forestry Incentive Program, established in 1974 and slated to end in 1995, funds timber production activities, some of which (e.g., tree planting) may enhance water quality. Over 3.9 million acres have benefitted so far in 49 states, with 70% in the southern states. Finally, the Stewardship Incentive Program, which began disbursing funds in 1992, supports a number of environmental protection activities, including stream bank stabilization, riparian buffer zones, and protection of native vegetation. As of the spring of 1992, about half of the states reported using Stewardship Incentive Program funds. Others were in the process of requesting them. The Agricultural Stabilization and Conservation Service administers the first three of these programs, but forestry aspects of the programs are facilitated by the USDA Forest Service in cooperation with state personnel. The Stewardship Incentive Program is administered by the USDA Forest Service, but the funds are disbursed with the assistance of the Agricultural Stabilization and Conservation Service. In addition to these four cost-share programs, the Federal Income Tax Reforestation Incentive Program provides credits for tree planting.

Summaries of state legislation and programs are provided by NCASI (1983), Cubbage et al. (1987), Guldin (1989, Appendix C), Essig (1991), and Brown et al. (1993). In table 13 and the following paragraphs, we provide a brief summary, as of spring 1992, of state approaches to control nonpoint source pollution from forestlands.

In the Southeast, all states have forestry BMP plans, and two states have grazing management plans, most of which employ voluntary practice guidelines (i.e., BMP's) to be implemented through training and educational programs (see table 13 for the states included in the southeast region). One state (Virginia) offers state-funded cost sharing (for agricultural BMP's that may apply in woodland areas). North Carolina, Florida, and West Virginia require the use of BMP's for certain road construction and silvicultural practices (Lickwar et al. 1990). Across the region, about 24 person-yr were

Table 13.—Number of states with programs and activities to control nonpoint source pollution on forest lands, as of spring 1992.[a]

Region[b]	Total	Silvicultural BMP's[c]			Grazing BMP's[d]	Financial incentives[e]	Implementation monitoring		Effectiveness monitoring (some activity)
		V	R	V/R			Some activity	Formal survey[f]	
Southeast	12	9	2	1	2	1	12	9	6
Northeast	11	4	4	3	1	2	9	3	6
N. Cent. & Great Pl.	8	7	0	0	1	5	3	1	2
Great Plains	6	2	0	0	0	1	4	1	1
Rocky Mountains	6	1	2	1	1	2	3	3	2
Pacific Northwest	3	0	3	0	0	0	3	2	3
Pacific Southwest	4	0	2	0	0	0	2	1	2
U.S.	50	23	13	5	5	11	36	20	22

[a] *This table summarizes a state-by-state table in Brown et al. (1993), which was based on phone interviews with personnel from forestry and/or environmental agencies in each state.*

[b] *Southeast: Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, Virginia, West Virginia. Northeast: Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, and Vermont. North Central and Great Plains: Iowa, Indiana, Illinois, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, Oklahoma, South Dakota, Texas, Wisconsin. Rocky Mountains: Colorado, Idaho, Montana, Nevada, Utah, Wyoming. Northwest: Alaska, Oregon, Washington. Southwest: Arizona, California, New Mexico, Hawaii.*

[c] *V = voluntary program of state approved BMP's; R = regulatory program of state approved BMP's, fines can be assessed for noncompliance; V/R = a combination of voluntary and regulatory approaches.*

[d] *Maine's program is regulatory; the others are voluntary.*

[e] *State-funded cost sharing or tax incentives.*

[f] *Formal periodic post-hoc survey of all or randomly selected sites meeting criteria for selection.*

devoted to nonpoint source pollution control programs in 1987, with a total budget of almost $1 million (table 14). That expenditure has likely increased with the additional effort that many states are allocating to monitoring, as discussed later.

In contrast to the Southeastern states where voluntary programs prevail, no one program type dominates in the Northeastern states (table 13). Of the 11 states in the region, 4 have regulatory programs, 4 have voluntary programs, and 3 use a combination of the 2 approaches. Of the three states with combinations, two (Massachusetts and New Hampshire) have regulatory programs of BMP's for riparian zones and voluntary programs for non-riparian forestry sites, while New York has a regulatory program for state-owned lands and a voluntary program for private lands. Two northeastern states (Maryland and New Jersey) offer tax incentives for using forest management BMP's, and Maryland has a state-funded cost-sharing program encouraging reforestation.

Nine of the 14 states in the North Central and Great Plains regions now have voluntary programs of state-approved BMP's for forestlands. Many of the states have relatively few forested areas, usually associated with farms and ranches, and thus have felt under less pressure than states in other regions to institute formal forestry BMP programs. Some states have relied on federal regulations where the forests tend to be federally owned. Only Iowa has a program of grazing BMP's, but Kansas is considering formulating them for riparian areas. Illinois has a state-funded cost-sharing program encouraging use of forestry BMP's, and Minnesota has a state-funded program emphasizing protection of riparian areas from livestock damage. In addition, Indiana and Illinois offer tax incentives for use of BMP's in woodland or forest areas. Wisconsin offers both tax incentives and cost sharing for maintenance of woodland through a formal management plan.

Four of the 6 Rocky Mountain states have nonpoint source pollution programs affecting forestlands. Colorado's program is voluntary, while Montana's is regulatory for riparian areas and voluntary elsewhere. Idaho and Nevada have regulatory programs, with Lake Tahoe Basin BMP's being more restrictive than those applying to other parts of Nevada. Two states offer tax incentives—Colorado for tree planting and Idaho for maintenance of forestlands on private property. Only Idaho has so far adopted grazing BMP's.

All three Pacific Northwest states have regulations for controlling of nonpoint sources of pollution from forest practices. In Alaska, a Forest Resources and Practices Act, passed in 1981, requires notification prior to harvesting operations and tries to prevent problems by advising use of BMP's. Alaska's program was strengthened in 1991, and more strict regulation is likely to result. In Washington, Forest Practice Rules and Regulations (pursuant to the 1974 Forest Practice Act) provide standards governing road construction, tree harvest, site preparation, chemical use, and reforestation. Written applications prior to operations are classed into one of five categories, with each category receiving different levels of evaluation. The Oregon Forest Practices Act of 1971 covers road construction, tree harvest, site preparation, chemical use, and reforestation. Practices are regulated on private and state lands.

In the Pacific Southwest, California and New Mexico have regulatory programs; Hawaii relies on strict land use planning requirements rather than BMP's; and Arizona is considering adoption of voluntary silvicultural and grazing BMP's. California's thorough regulatory program involves a combination of legislation, administrative regulation, active enforcement, and licensing of professional foresters and timber operators (Yee 1987). Every timber harvest in California must include a timber

Table 14.—Personnel and budgets for state forestry-related nonpoint source pollution control programs in the Southeast, 1987.[a]

State	Full-year equivalent agency employees		State agency budgets	
	Water quality	Forestry	Water quality	Forestry
Alabama	1.0	0.5	$ 10,000	$10,000
Arkansas	0.0	2.5	0	64,000
Florida	4.7	2.0	211,000	60,000
Georgia	1.0	1.3	0	50,000
Kentucky	0.0	0.5	0	50,000
Louisiana	0.0	0.0	0	0
Mississippi	0.0	0.0	0	0
N. Carolina	1.0	1.5	120,000	40,000
Oklahoma	0.0	1.8	0	80,000
S. Carolina	0.0	0.2	0	5,000
Tennessee	0.0	0.0	0	0
Texas	0.0	0.0	0	0
Virginia	2.0	4.0	85,000	150,000
Total	9.7	14.3	426,000	509,000

[a]*Source: Lickwar et al. (1990).*

harvest plan that is reviewed by an interdisciplinary review process. Once a permit is granted, the Department of Forestry has enforcement responsibilities to ensure compliance with a wide range of regulations (including water protection). This elaborate process was estimated to increase the stumpage cost by about 5-10%. Cooperation between the state and USDA Forest Service has led to an intensive program for maintaining water quality on National Forests, including personnel training, development and refinement of BMP's, a handbook on BMP's, and implementation and monitoring of BMP's in forest operations (Leven et al. 1987).

The Forest Service BMP's are designed to be flexible because water quality problems vary substantially among California's forests. Leven et al. (1987) reported 98 official BMP's grouped in 8 categories from road construction to vegetation management to grazing impacts. The BMP's related to road construction include guidelines on constructing roads of minimum length that conform to the terrain, with well-designed drainage. The rules also require buffer strips along streams, directional falling of trees away from streams, and no physical impact on stream channels (Skaugset 1987). The application of BMP's proceeds in four phases: feasibility, site-specific assessment, application of BMP's, and monitoring. During 5 yr in the early 1980's, about $3.3 million was spent to correct nonpoint source pollution problems resulting from deteriorated watersheds in National Forests in California, but the estimated backlog of rehabilitation projects was $57 million (including $37 million simply for erosion problems).

In addition to state programs, many local ordinances have been passed by counties, townships, and municipalities. Martus et al. (1991) identified 377 local ordinances that regulate forestry activities in the United States, with 72% of them in the Northeastern states. About three-quarters of the ordinances were enacted in the past 10 yr, and nearly half are less than 5 yr old.

Best Management Practices

Whether voluntary or regulatory, state and local programs typically rely on a set of land management practices that land managers are encouraged to follow. These practices are often called best management practices (BMP's), but some states use "acceptable management practices," "forest practice rules," or other terms. As Wilkinson and Anderson (1985:220) report, EPA regulations define BMP's as

> those methods, measures, or practices to prevent or reduce water pollution and include but are not limited to structural and nonstructural controls, and operation and maintenance procedures. BMP's can be applied before, during, and after pollution-producing activities to reduce or eliminate the introduction of pollutants into receiving waters. Economic, institutional, and technical factors shall be considered in developing BMP's (40 C.F.R. 35.1521-(4)(c)(1), 1984).

On forested land, the following BMP's are sometimes used to minimize or prevent nonpoint source pollution from timber harvest: (1) buffer strips along perennial and intermittent streams, where logging is prohibited or limited to selective removal of high-value or undesirable trees; (2) prohibition of skidding over streams, except over approved culverts or bridges, (3) supervision of logging by a qualified forester or engineer; (4) division of timber sales into more easily administered blocks that are harvested one at a time; (5) prohibition of disposal of tops or slash near streams; (6) proper location of haul roads, skid trails, and log landings to avoid soil loss; (7) retirement of skid trails and haul roads after logging; (8) installation of water bars and other erosion control and drainage devices where necessary; (9) seeding and other efforts to maintain vegetative cover; and (10) prohibition of logging during excessively wet periods (Lynch et al. 1985).

Officially designated BMP's for rangeland are less common than those for forests, but more states, in the West, are now taking steps to specify rangeland BMP's. Rangeland BMP's emphasize limiting grazing intensity by controlling (1) livestock numbers, (2) the timing of livestock use, and (3) livestock distribution (with fencing, herding, salt placement, and water development) (Chaney et al. 1990). Implementation of rangeland BMP's often focuses on riparian areas, where the impacts of grazing on water quality are potentially greatest. Other practices aim at improving rangeland vegetation by seeding and at assuring careful brushland management and prescribed burning.

Undoubtedly, BMP's can be designed that will contain the effects of harvest, grazing, and other activities to within acceptable limits. Some careful studies implementating BMP's (e.g., Lynch and Corbett 1990 on Pennsylvania's silvicultural BMP's) have demonstrated the effectiveness of BMP use in protecting water quality. However, the fact that using of a certain set of BMP's is effective in one location does not guarantee that those BMP's will be effective in a different location. The soils and their slopes, weather patterns, and several other factors must be considered in the selection of the most effective site-specific BMP's.

Whitman (1989) suggests that in some conditions, such as areas of steep unstable slopes, BMP's alone are insufficient to control sediment loss to within acceptable limits and that in such conditions the land management planning process should be used on public land to preclude such areas from harvest. His suggestion assumes that BMP's cannot be used directly to exclude some areas from harvest, an assumption that may unnecessarily restrict the purview of the BMP process. BMP's for restricting harvest along stream buffer zones are now common; perhaps the same concept of exclusion could be extended to areas of steep slopes with unstable soils.

Cases where BMP implementation fails to achieve water quality objectives have led to conflict, which sometimes ends up in court. The decision in the so-called *Blue Creek* case involving National Forest land in California was that water quality standards could constitute judicially enforceable constraints on land management. Anderson (1987:605) summarized the Ninth Circuit Court of Appeals' 1986 decision: "Even if all applicable BMP's are followed, a given project or group of projects may be illegal under the CWA [Clean Water Act] if the evidence indicates that the resultant pollution will exceed state standards." However, after the Blue Creek case, the EPA clarified the role of BMP's in nonpoint source pollution control and the relation of BMP's to water quality standards. The EPA (1987) guidelines state in part:

> Once BMP's have been approved by the State, the BMP's become the primary mechanism for meeting water quality standards. Proper installation, operation and maintenance of State approved BMP's are presumed to meet a landowner's or manager's obligation for compliance with applicable water quality standards .... For proposed management actions, BMP's designed and implemented in accordance with a state approved process will normally constitute compliance with the CWA.

The guidelines go on to emphasize the iterative nature of BMP specification (involving implementaiton, monitoring, and subsequent adjustment of BMP guidelines) and the role of standards as a base against which the effectiveness of BMP's are to be measured (Rector 1989). Thus, the difficulty of specifying BMP's to precisely meet standards was acknowledged, the importance of continually upgrading BMP guidelines was highlighted, and the focus of compliance on BMP implementation was reinforced.

Implementation and effectiveness monitoring

Even if BMP's are appropriately specified for the site, they must be implemented. And the effectiveness of their use must be checked to allow reassessment of BMP requirements. Thirty-six of the 50 states reported performing implementation monitoring activities (table 13). States use different procedures for encouraging and checking on compliance (NCASI 1988). Some states, especially those with regulatory programs, rely on visits by state forestry personnel to sites while management practices such as harvest and road construction are in progress. Because ongoing inspection of forest management in progress is expensive, inspectors may only visit the most important sites. Twenty states employ a formal survey of randomly selected recently managed sites (table 13), while others use a less formal inspection of sites or an ad hoc inspection of sites suspected of not being in compliance. Some states include federal lands in their formal surveys, but most leave that to the federal agencies.

Monitoring also occurs where a contract or agreement between the state and a private party requires BMP implementation. This may occur where landowners benefit from financial incentives or where contracts for harvest on state land contain BMP clauses.

The USDA Forest Service now distinguishes between two kinds of monitoring that we group here under effectiveness. First, "effectiveness monitoring" determines whether implemented practices performed as expected. Such monitoring does not necessarily measure water quality. For example, if a practice is designed to reduce sediment delivery to a stream, effectiveness monitoring would inspect on-slope sediment movement. Effectiveness monitoring may use quantitative or qualitative methods. Second, "validation monitoring" determines whether water quality standards are met, and whether water quality prediction models are accurate. Quantitative methods are needed here (Warren Harper, USDA Forest Service, personal communication).

Effectiveness of BMP's implemented on site can basically be checked in two ways: qualitatively by trained professionals during onsite inspection, or by quantitative measurement. Qualitative checking can be accomplished informally or preferably via a formal survey of randomly selected sites, perhaps in the course of a compliance survey. Qualitative checks may miss difficult-to-observe levels of suspended sediment or other constituents that might be found through analyzing water quality samples. Quantitative measurement can include downstream water quality sampling, bedload monitoring, and biological monitoring, as well as on-land monitoring of soil movement. Careful quantitative measurement is preferable to qualitative judgments, but its high cost often limits such measurement to a few carefully selected sites. Twenty-two states reported performing some effectiveness monitoring activities (table 13); five of these employed some quantitative monitoring.

Formal surveys of BMP implementation indicate a range of compliance and effectiveness. Several such studies are summarized here.

1. Florida, which has a largely nonregulatory BMP program (Lickwar et al. 1990), has conducted biannual

compliance checks of selected sites since 1979. Sites selected for investigation were subject to a silvicultural operation (e.g., harvest, site preparation, a regeneration activity) during the previous 2 yr and are located within 300 ft of either a perennial or intermittent stream or lake of at least 10 acres. Eighty-five questions are answered at each site by the county forester, some of which focus on effectiveness of BMP use. The survey concludes with an overall judgment of whether "there was (generally) good compliance with 208 guidelines." In the 1989 survey, 94% of the 128 sites surveyed were judged as generally in compliance (Conner et al. 1989). Overall compliance was 89% in 1987 and 84% in 1985. Additional efforts were recognized as needed to "sensitize equipment operators on the proper use of equipment on more erodible soils" and to improve stabilization of stream crossings (Conner et al. 1989:8).

2. Georgia, another nonregulatory BMP state, recently completed its first large scale compliance survey (Georgia Forestry Commission 1991). The survey focused on BMP's dealing with five types of actions: road construction, harvest, site preparation, reforestation, and fire control. A total of 345 sites where a forestry operation had been completed within the previous 6 months was surveyed. Compliance across all 5 types of BMP's was 86%. Compliance ranged from 69% for road construction BMP's to 96% for reforestation BMP's. Ninety-five percent of the length of stream banks and channels within the survey sites was judged to be "intact and unimpaired." The report concluded that "current BMP's appear to be sufficient in protecting water quality when implemented," but that "it may be necessary to modify some BMP's, be more site specific, and address changes in equipment and technology" (Georgia Forestry Commission 1991:23).

3. In South Carolina, Hook et al. (1991) evaluated BMP compliance on 100 recently logged areas selected on aerial photos to represent a wide range of wetland or riparian site types and a range of harvest area sizes, and to be representative of the state's forestland ownerships and landscape types. The 7 team members from agencies, academia, industry, and a conservation group visited the sites during a 5-month period in 1990. Members recorded their subjective assessments. Nearly all of the sites were on industry or private land, and 61 were on private holdings of less than 1000 acres, or about 400 ha. Overall compliance was 95% on industry land, 86% on private holdings of greater than 1000 acres, and 78% on private holdings of less than 1000 acres. Across all ownerships, compliance varied from about 50% for streamside management zone BMP's along navigable streams to 90% for log deck BMP's. Only 56% of the landowners indicated that they were aware of the voluntary BMP's; lack of awareness was more common for the small forest owners, who were less likely to contract for the services of a professional forester. In another area of South Carolina with 177 harvested sites, Adams (1992) found an overall BMP compliance of 85%, with compliance ranging from 42% for road stream crossing BMP's to 98% for log deck BMP's.

4. The Virginia Department of Forestry attempts to obtain an inspection of all harvested forest areas of five acres or more by either a Department employee or a participating industry or consulting forester. In 1990, based on inspection of over 1000 sites for use of voluntary BMP's, compliance was judged to vary from 84% for skid trails to 98% for site preparation (Virginia Department of Forestry 1991). Compliance with haul road layout, haul road stabilization, landings, and streamside management BMP's was all above 90%.

5. Irland (1985) reported the results of two extensive field surveys of commercially harvested forests in the Northeast. First, in Connecticut a survey conducted in the late 1970s of 2100 ha of harvested forests (in 80 separate sites) revealed that severe gullying developed on 15% of the logged units. Some gullies were as deep as 3 m. Skid trails crossed streams a total of 141 times on the 80 units. Second, a study of 56 harvesting operations in Maine in 1980 found that about 50% of the units showed substantial amounts of erosion or sedimentation. Most of the problems related to inadequate water control on logging roads. The impacts documented in these extensive surveys suggest that operational practices may have greater effects on water quality (particularly sediment loads) than the impacts documented in more intensively studied watersheds.

6. Brynn and Clausen (1991:143) found that compliance with Vermont's timber harvest "acceptable management practices" varied from 0 to 98% depending on the practice at the 78 silvicultural operations they investigated. Postharvest water body sedimentation was above "background levels" at 46% of the sites, but "heavy sedimentation" occurred at only 9% of the sites. The authors suggested that "future research should focus on the impact of timber harvesting operations as conducted under economic constraints rather than unrepresentative research conditions," and they recommended that BMP's "should accurately reflect the economic and technical constraints of ... timber harvesting while adequately protecting water resources from degradation."

7. Texas' first systematic compliance survey investigated recently harvested sites in east Texas (Texas Forest Service 1992). An original sample of 257 sites was selected in a stratified (by county and ownership) quasi-random manner, but time constraints limited onsite inspection to 162 sites. Two foresters jointly visited all sites from mid 1991 to mid 1992, completing a 73-question checklist at each. Overall compliance was rated as good or excellent on two-thirds of the 162 sites and fair on another 22% of the sites. Overall good or

excellent ratings were assigned to 80% of the public land sites, about 73% of the industry and large nonindustrial private sites, and 56% of the small nonindustrial private sites. Compliance was highest where a forester was involved and where the landowner and logger were familiar with BMP's. The most common problems were associated with stream crossings. Based on qualitative assessment, the report concluded that the BMP's were effective in controlling nonpoint source pollution when BMP's were implemented.

8. BMP use is mandatory in Idaho. An interdisciplinary team audited the impacts of forest management on water quality from 40 projects across Idaho (Harvey et al. 1988). Ten projects were selected from each of the following ownerships: National Forests, Idaho Department of Lands areas, forest industry land, and private nonindustrial land. The audit team included people with expertise in fisheries biology, hydrology, road construction, and water quality from the USDA Forest Service, state agencies, and private industry. The team examined whether BMP's were implemented, whether they were effective, and whether any problems were more common on a particular type of land ownership. Compliance with BMP's was high on public and industrial lands, averaging about 95%. Nonindustrial private lands complied with BMP's about 86% of the time. Compliance with BMP's led to no stream sedimentation problems in 99% of the cases, whereas noncompliance led to sedimentation problems in 70% of the cases.

In 1991, the Idaho Department of Lands assessed BMP compliance and effectiveness for 40 timber sales (23 on state land and 17 on private land). Sales were selected by using a variety of criteria and do not represent a random sample. The 40 sales were located within a half-day's drive of an area office, and the private land sales were all in areas draining into "stream segments of concern." Five of the state land sales and eight of the private land sales had some degree of noncompliance resulting in minor water quality impacts. The assessment concluded that "when rules/BMP's are implemented they are effective in minimizing impacts to beneficial uses" (Colla 1992).

9. In Montana, 44 recently harvested sites were surveyed in 1990 by 6-member interdisciplinary teams who rated up to 58 BMP's at each site for compliance and effectiveness (Schultz 1990). The sites were chosen randomly from among a set of sites that met certain criteria, including minimum proximity to a stream and minimum size of harvested area. Two-thirds of the sites were "high hazard" sites, as determined based on slope, erodibility, and riparian proximity. Regarding BMP compliance across all sites, 78% of the BMP applications met all requirements and 14% were only minor departures, with the remaining 8% being major departures. However, for the 9 BMP's most important for protecting water quality, only 53% of the applications met all requirements, 29% were minor departures, and 18% were major departures. Regarding effectiveness across all sites, 80% of the applied practices were rated as providing adequate protection and 11% as potentially causing only minor impacts, with the remaining 8% potentially causing major impacts. However, among the 9 most important BMP's, only 58% of the actual applications were rated as providing adequate protection, with 19% potentially causing minor impacts and 23% potentially causing major impacts.

10. In a 1980 assessment of randomly selected sites in Washington, forest practices were in compliance with established regulations 80% of the time (Sachet et al. 1980). Compliance led to almost no water quality problems, but water quality impacts occurred in about 70% of the noncompliance cases. The most recent survey occurred in 1991, of 191 randomly selected application sites throughout the state where harvest, road construction or maintenance, or chemical use occurred from 1987 to 1991 (TFW Field Implementation Committee 1991). The sites were divided among four evaluators, who were assisted in some cases by other experts. Some of the sites had received visits from state personnel before (31%) or during (18%) the operations. The survey found that while 37% of the applications had differences between what was done and what was stated on the application, only 14% of those (5% of the total) did not meet or surpass the regulations. Only 1% of the applications resulted in damage or potential damage to the public resource.

11. A 1989 assessment of 5,204 operations in Oregon, selected by a priority ranking, found that 97% of the operations were in compliance with state forest practice rules (Oregon Department of Forestry 1990). Of the 190 citations issued for noncompliance, 61 were for failure to notify the state forester and 31 were for violations of written plans, with the remainder dealing with onsite actions such as harvesting and road construction. Other recent assessment efforts in Oregon have dealt with specific issues, such as herbicide use and riparian areas. For example, in 1989 and 1990, water quality samples were taken from 50 herbicide application units in western Oregon. The applied herbicide was not detected in 43 of the samples, and all detected herbicide levels were below research-based monitoring standards (Oregon Department of Forestry 1992).

12. In California, a 4-person multidisciplinary team evaluated compliance with and effectiveness of BMP's on 100 harvest units on nonfederal land selected on a stratified random basis (SWRCB 1987). Implementation of BMP's was variable, but protection measures were generally effective in about 60 of the 100 projects. Where protection was insufficient and resources were placed "at risk," the actual impacts on streams were generally minor, although the impacts at some sites

were moderate to major and a few were judged to be severe. Impacts on streams were generally minor when procedures outlined in timber harvesting plans were followed. The team concluded that "...noncompliance [with forest practice rules] was the single most important impediment to achievement of adequate resource protection" (SWRCB 1987). In 1988, 7,578 onsite inspections by Department of Forestry staff to determine compliance of timber operations with California's forest practice rules found 481 violations (6%), with construction of water-breaks, treatment of slash, watercourse protection, and road maintenance being the most common problems (CDF, 1988). Also, see Knopp et al. (1987) for an examination of the adequacy of BMP's in protecting water quality in the Six Rivers National Forest.

Most states are now performing some sort of compliance survey, and formal surveys of randomly selected sites is the preferred approach. There has been a dramatic increase in the number of states performing formal surveys of BMP compliance. Encouraging results from such surveys are now generally considered to be necessary justification for continuing with voluntary (as opposed to regulatory) nonpoint source pollution control programs. Effectiveness surveys are also becoming more common, with qualitative surveys of randomly selected sites being the most common approach. The obvious trend among the states is toward a more concerted monitoring effort, employing periodic surveys using well-established survey methods.

Overall, it appears that compliance with BMP's is generally high and gradually improving[6] and that water quality is usually within standards where BMP's are implemented. However, cases of noncompliance persist and water quality problems were often associated with such noncompliance, suggesting that continued efforts are needed to ensure BMP implementation. Because the bulk, if not all, of the onsite costs of BMP implementation are borne by the landowner, while the benefits typically accrue to aquatic organisms and downstream water uses, noncompliance may sometimes seem to landowners like an attractive alternative, especially in voluntary states. Thus, compliance and effectiveness monitoring must be an ongoing activity, and instituting a regulatory program must remain a realistic possibility.

Are BMP's the best approach?

The goal of water quality protection programs is to meet standards in the most cost-effective way. BMP's are an administrative approach to reaching this goal. Specifying BMP's to cost effectively reach water quality standards requires an understanding of the complex relations between land disturbance and downstream water quality, as well as of the costs of alternative practices. The complexity arises in part from the difficulty of (1) distinguishing among the individual causes of water quality degradation in a watershed to know the contribution of each area and land practice (a formidable task for "nonpoint" source pollution) and (2) separating natural from management-caused water quality degradation in the context of a variety of weather events. Monitoring of water quality is essential to understand the relations between land disturbance and water quality. By observing the effect over time of precipitation events on water quality downstream of disturbed and undisturbed areas, scientists and land managers can improve their understanding of these relations. This improved understanding can then be used to reassess BMP guidelines so as to more cost effectively reach water quality goals in the future. This iterative process of BMP specification, use, monitoring, and then fine-tuning of BMP specifications for future applications is the key to cost-effective BMP use and effective water quality protection. It relies heavily on gradually improved understanding of the effect of site-specific land management controls on downstream water quality.

Some have called for sufficiently extensive monitoring programs that compliance could be judged directly in terms of meeting water quality standards rather than in terms of applying required BMP's. With achievement of water quality standards as the criterion, landowners would be free to choose the most cost-effective practices on a site-by-site basis to meet prescribed water quality standards for the larger watershed in which the sites are found. However, this idealized approach would only be workable with sufficient water quality monitoring to isolate the specific land area source of the problem and to determine whether the water quality degradation would have happened even in the absence of the land disturbance. Providing such detailed information would require continuous long-term monitoring of both treatment and control sites at many points along the stream network. Applying a comprehensive monitoring program like this over the many areas subject to harvesting and heavy grazing would be very complex and costly. Another problem is that the water quality impacts of land disturbances may not occur until extreme weather conditions develop, which may happen several years after the disturbance. The practical solution has been to (1) prescribe land management practices (i.e., BMP's) that careful studies and professional judgment indicate will control nonpoint source pollution to within standards in most cases, and then (2) to reassess BMP guidelines as new information becomes available. Although the goal of the water quality program is to keep water quality within the standards, the

[6] It should be mentioned that states that have performed formal surveys of BMP implementation may tend to be those that have taken a more proactive stance in explaining the practices to forest managers and operators and in promulgating their use.

immediate objective of the program then becomes the implementation of prescribed BMP's.

Water quality standards are cost effective when they are met accurately, without over- or under-constraining land management. The cost of overconstraining land management is in the waste of resources and consequent loss of income on the part of the landowners. The potential cost of underconstraining land management is in the effect of poor water quality on aquatic organisms and downstream water users.

Common procedures for checking BMP compliance and effectiveness may tend to limit the cost effectiveness with which water quality standards are met. Compliance and effectiveness surveys usually focus on whether or not the goal was met, not on the accuracy with which the goal was met. Exceeding the standard tends to be regarded as a bonus of BMP use, without regard to the cost of implementation. Where BMP implementation is costly and exceedance of the standard is not of comparable value to the cost of exceedance, evaluations of effectiveness of BMP's should measure for over- and underachievement, and future BMP requirements should be adjusted up or down to allow more cost-effective future achievement of the water quality standards.

The cost effectiveness with which BMP's meet water quality standards also depends on how well the BMP's were chosen for a given condition. The more carefully BMP's are tailored to the site-specific conditions, the more likely that they will cost effectively reach their stated goals. Because the professional expertise to carefully select BMP's is costly, BMP's are often specified for large geographical areas (such as counties or multicounty regions), although nonpoint source pollution in specific sites within the larger area may be more inexpensively controlled with one set of BMP's than another. This is not the fault of the BMP approach—rather, it is a matter of how BMP's are specified. The more carefully they are specified for a given site, the more cost effectively the water quality standards will be met, all else equal.

BMP specification must, of course, deal with the complex area of risk. The extent of water quality degradation resulting from land disturbance depends on *when* unusual precipitation events occur. There will be some risk that a severe event could occur soon enough after the land disturbance to cause serious increases in water quality degradation, over and above the background degradation (without the disturbance) that such an event would cause. BMP specification should somehow incorporate an understanding of these risks and reflect a judgment about the level of risk that society is willing to accept.

Costs to the landowner are not the only costs of BMP implementation. Specification of site-specific BMP's by a trained professional, and periodic adjustment of the level of BMP implementation to more accurately attain the water quality goals, can also be costly. These costs should be compared with the costs of overconstraining land management practices to help determine the most efficient level of professional assistance needed in carrying out a BMP program. However, as a general rule, the availability of well-qualified personnel at the field level is probably the most cost-effective approach to meeting water quality standards.

Benefit-cost Comparison of Water Pollution Controls on Forestland

The preceeding discussion of BMP programs focused on the cost effectiveness with which BMP's are specified and implemented to assure that water quality is within water quality standards. That discussion assumes that water quality standards are to be met regardless of the costs. This chapter steps back to compare, to the extent possible given existing literature, the benefits and costs of BMP use. This benefit-cost comparison adopts the perspective of economic efficiency, rather than the more limited perspective of cost effectiveness. Economic efficiency focuses on both benefits and costs. It attempts to do so regardless of to whom the benefits and costs accrue, and regardless of whether the benefits and costs are for goods and services traded in established markets.

The cost of adhering to BMP's may turn a financially profitable timber sale or other operation into a money loosing endeavor, leading to pressure to relax the BMP requirements. However, the effect of BMP constraints on financial returns is not sufficient justification for relaxing BMP specification. It may be that when the true social costs of a sale are tallied, they exceed the benefits, and the sale should not go forward. However, it is also feasible that at some sites the costs of BMP use exceed the benefits of that use. That is, the costs of BMP implementation (increased road construction cost, decreased harvest, etc.) may exceed the cost of the onsite and downstream damage that the BMP's would avert. In this case, it may be reasonable to relax the BMP specifications to the level where the benefits from their use equal their cost.

Section 319 of the Clean Water Act requires the state reports to describe "the process ... for identifying best management practices ... and to reduce, to the maximum extent practicable, the level of pollution ..." (33 U.S.C. 1329(a)(1)(C)). It is not clear what criteria should be used to determine practicability. In particular, should economic considerations enter in determining "practicable" level of control? Because the act also encourages collection and sharing of "information concerning the costs and relative efficiencies of best management practices for reducing nonpoint source pollution" (33 U.S.C. 1285 (l)), there is some suggestion that economic data may be relevant in decisions about BMP specification.

Identification of the physical effects of water quality degradation, and estimation of the social costs of those effects in monetary terms, is a difficult task likely to yield only rough approximations of the true values. Nevertheless, even a rough comparison of such costs of water quality degradation with the costs of avoiding the degradation might provide useful input toward decisions about control efforts. We present a rough comparison here, focusing on erosion, which is the principal water quality effect of silvicultural and related construction activities.[7]

The offsite costs of erosion from various causes have been estimated for specific locations by many authors. Clark et al. (1985) summarized many of these estimates and extended them to the entire area of the 48 conterminous states, and Ribaudo (1986) updated and reorganized these estimates. For sediment and associated categories of nonpoint source pollution, Ribaudo estimated the annual damage cost from erosion of various causes to be from \$4.4 billion to \$16.1 billion, with a best estimate of \$7.6 billion (adjusted to 1985 dollars using the GNP deflator). As the range suggests, the authors recognize the considerable difficulty of estimating such damages. And the difficulties of extrapolation of site specific studies to other areas is an acknowledged problem (Devousges et al. 1992). In any case, the magnitude of total impact is impressive. Damages to recreation and fishing account for about 40% of the total best estimate; damages to water storage and conveyance facilities, ditches and canals, and navigable channels sum to 30%; damages to municipal and industrial users are 17%; and flood-related damages are 13% (table 15).

Ribaudo (1986) disaggregated the table 15 estimates to regions of the United States, and expressed the damage estimate on a per-unit of sediment basis. Regional estimates range from \$0.57/Mg for the Northern Plains states to \$6.45/Mg for the Northeast states (table 16). Higher costs per Mg were associated with important fishery resources and heavily populated areas. Across all 10 regions, the damage estimate is \$1.60/Mg. Erosion source areas for the damage estimates of table 16 included cropland, pasture, rangeland, forests, construction sites, mines, quarries, and stream banks. Neither Clark et al. (1985) nor Ribaudo (1986) specified how much of the total damage is attributable to forests and rangelands. Table 7 lists the average sediment discharges from the different types of land, but costs per Mg are not necessarily proportional to discharge rates. The costs per Mg should be higher for those land types where the sediment is more likely to carry other constituents of water pollution, such as pesticides, salts, and toxics. Such constituents are more likely to be attached to soil leaving farms, mines, and urban areas than to erosion from forests and rangelands.

Ribaudo's estimates of damage per Mg indicate the benefit of reduced erosion and associated contaminants, assuming a linear damage function. The benefit of reduced erosion can be compared with estimates of the costs of controlling erosion on forestlands to allow

[7] For a national benefit-cost comparison focusing on point sources, see Freeman (1982).

Table 15.—Annual offsite damages from erosion for the 48 conterminous states (1985 dollars).[a]

Damage category	Million dollars	Percent of total
Freshwater recreation[b]	2,016	27
Marine sport fishing	591	8
Commercial freshwater fishery	59	1
Commercial marine fishery	378	5
Water storage facilities[c]	1,171	15
Dredging navigable waters	726	10
Flooding[d]	948	13
Drainage ditches and culverts[e]	228	3
Irrigation canals[f]	114	2
Municipal and industrial water use[g]	1,315	17
Irrigated agriculture[h]	30	<1
Total	7,564	100

[a] *Source: Ribaudo (1986), who relied heavily on Clark et al. (1985).*
[b] *Damage to fishing, boating, and swimming.*
[c] *Costs for lost storage capacity (where replacement is infeasible), replacing lost storage capacity, and dredging.*
[d] *Damage from increased flood heights due to channel aggradation; increased flood volumes due to sediment loads, direct sediment damages, and reduced agricultural activity.*
[e] *Based on the cost of keeping them clear.*
[f] *For sediment removal and increased weed control.*
[g] *Based on damages and on the cost of removing sediment and associated contaminants to acceptable levels.*
[h] *Based on costs of salinity.*

Table 16.—Annual offsite damage from soil erosion, by region (1985 dollars).

Region	Damage from all sources (millions of dollars)	Erosion from all sources (millions of Mg)	Damage per Mg (dollars)
Appalachian	566	446	1.27
Corn Belt	991	894	1.11
Delta states	517	216	2.40
Lake states	553	167	3.53
Mountain states	868	925	0.94
Northeast	1099	171	6.45
Northern Plains	351	619	0.57
Pacific	1441	617	2.34
Southeast	343	230	1.48
Southern Plains	837	452	1.85
Total	7564	4736	1.60

Source: Ribaudo (1986).

a rough benefit-cost comparison. Erosion control costs have been estimated in several forest areas, including those of the following five studies. All costs have been adjusted to 1985 dollars using the GNP deflator.

1. Hickman and Jackson (1979) estimated the costs to timber owners of reducing erosion from the roughly 150,000 ha of commercial forest land in Cherokee County in northeast Texas. Costs were in terms of reductions in income resulting from restrictions on site disturbances. Erosion was estimated for the 18 relevant soil types using the universal soil loss equation (USLE), but no attempt was made to determine what portion of the soil loss would be transported to streams. On average, these costs were about $10/Mg for initial reductions and higher thereafter. These costs are over 5 times the Southern Plains states' damage estimate (table 16) of $1.85/Mg. Expressing the cost in terms of units of sediment reaching the stream, rather than in terms of onsite erosion, would increase the estimate of cost per Mg and further weaken the case for the erosion control practices analyzed.

2. Miles (1983) compared the costs of implementing 6 practices (water bars, broad-based dips, buffer strips, culverts, skid trail and landing design, and seeding of roads and landings) on 2 timber sales, a 40 ha area in Minnesota, and a 52 ha area in Michigan.[8] Using the USLE to estimate erosion and sediment loading factors to estimate sediment delivery to the stream, and assuming that the 6 practices would completely avoid the harvest effects, Miles concluded that the cost of the avoided sediment was $69/Mg for the Minnesota site and $39/Mg for the more erodible Michigan site. These costs are considerably above the Lake and Corn Belt states' average damage estimates.

3. Ellefson and Weible (1980) estimated the cost of implementing BMP's during a timber harvest of a 42 ha area in Minnesota. The cost was about $26.50/ha for filter strips, seeding, and improved skid trail design and implementation. Given the Lake states' damage estimate of $3.53/Mg, the BMP's would have to prevent about 7.5 Mg/ha of soil reaching the stream for benefits to match costs.

4. Lickwar et al. (1992) estimated a cost of about $29 ha for implementing currently required BMP's on 22 timber sale areas in 3 southeastern states. Given the Southeast states' damage estimate of $1.48/Mg, the BMP's would have to avoid about 20 Mg/ha of soil reaching the stream.

5. Olsen et al. (1987) estimated costs of implementing proposed Oregon forest practice rules on a representative 541 ha industrial forested watershed in the Oregon Coast Range. The rules would increase restrictions on harvesting and related activities in riparian zones to improve soil stability and protect habitat. The least expensive option they evaluated essentially was incorporated as BMP's that became required by state law in 1987, shortly after the study was completed. Costs of this option were in terms of increased road and harvesting expenses and decreased harvest volume. Depending on the size timber on the site, the cost of implementing these restrictions varied from $250 to $595/ha.[9] Given the Pacific states' damage estimate of $2.34/Mg, the restrictions would have to avoid from 100 to 250 Mg/ha (depending on timber size) of soil reaching the stream if the implementation cost were to be completely covered by offsite water quality benefits.

The simple benefit-cost comparisons are summarized in table 17. The Texas study by Hickman and Jackson (1979) directly estimated onsite costs to timber owners per Mg of avoided erosion. Costs were considerably higher than the offsite benefits. This study evaluated alternative harvest and site preparation practices rather than typical BMP's. It presented results on an average annual basis for the county as a whole; therefore, it is not directly comparable to the other studies, which emphasize BMP's and erosion during and shortly after harvesting activities. The costs estimated by the other studies suggest that, for offsite benefits to equal onsite costs, erosion reaching the stream would have to be from 7.5 Mg/ha (for the Michigan study by Ellefson and Weible 1980) to 20 Mg/ha (for the Southeast areas studied by Lickwar et al. 1992) to 107 Mg/ha or greater for sites with larger timber (for the Oregon study by Olsen et al. 1987). In the studies summaried in the previous chapters, the short-term effects of harvest and related activities on sediment loss ranged from only about 0.05 Mg/ha/yr for some Colorado sites to from 4 to 14 Mg/ha/yr for most of the Southeast sites and 13 Mg/ha/yr for an Idaho site.

These rough benefit-cost comparisons suggest that the cost of avoiding stream sedimentation and associated water quality degradation on forestland often exceeds the offsite benefits of doing so. However, this suggestion must be qualified. The following factors support a more positive view of use of the practices, at least in some areas and to some extent in any given area: (1) The damage estimates are averages over large areas. Some site-specific damages (such as bridge failures) may significantly exceed these averages. (2) The damage estimates do not include effects on the downstream ecosystem (except for the associated impact on fishing value). Economic studies indicate that the public assigns considerable value (called "existence" or "intrinsic" value) to maintaining good water quality (Fisher

[8] Ellefson and Miles (1985) estimated the cost of these 6 forest practices for 18 timber harvests on 9 National Forests in 5 Midwest states, including the two mentioned here. However, they do not list the sizes of the sale areas, so we could not put their cost estimates on a comparable basis to the other studies.

[9] Another, less detailed, Oregon study (Garland 1987) of a similar level of additional riparian zone protection estimated a cost of $1240/ha.

Table 17.—Comparison of offsite benefits to onsite costs for forest erosion control practices (1985 dollars).

Study	State	Onsite costs ($/ha)	Onsite costs ($/Mg)	Offsite benefit ($/Mg)[a]	Erosion avoided to break even (Mg/ha)
Hickman and Jackson (1979)	TX	na	10+	1.85	na
Miles (1983)	MN	99	69	3.53	283
	MI	101	39	1.11	91
Ellefson and Weible (1980)	MN	26.50	na	3.53	7.5
Lickwar et al. (1991)	AL,GA,FL	29.00	na	1.48	20
Olsen et al. (1987)	OR	250+	na	2.34	107+

na = not available.
[a] From table 16.

and Raucher 1984). (3) The damage estimates do not include onsite costs, such as long-term loss of soil productivity.[10] (4) The damage function is not necessarily linear; initial reductions in water pollution may be worth more than the average per-unit reduction. (5) The cost function is not necessarily linear. Hickman and Jackson (1979) and Miller and Everett (1975) found that the marginal cost of reducing soil loss increased as additional erosion was controlled. Initial reductions in erosion will typically be less costly than the average costs listed above.

Conversely, the following concerns reinforce a skeptical view of BMP's on some forest land: (1) The costs listed are onsite costs and do not include the agency costs to inform forest owners about BMP's, to administer a nonpoint source pollution program, and to monitor compliance with BMP's. Lickwar et al. (1990), for example, found that in 1987 the 13 southern states spent about $935,000 on such forestry-related activities (see table 14), and at the time only one state (Florida) was regularly monitoring and enforcing BMP's. Several states indicated that their activities would increase after 1987. (2) Costs per Mg of erosion from forestland are likely to be lower than those for many other types of land because fewer other water quality contaminants are attached to soil from forestland than to soil leaving farms, cities, etc. (3) The damage function is not necessarily linear; initial reductions in water pollution may be worth less than the average per-unit reduction. For example, Ribaudo (1986) suggests that reductions in erosion do not significantly improve fish habitat and recreation quality until the sediment level falls below some threshold.

This benefit-cost comparison is not precise enough for site-specific recommendations about the use of BMP's or for general conclusions about the economic efficiency of BMP implementation. First, the regional average estimates of benefit received from water quality protection are rough at best. Second, the benefit estimates are especially general, each covering a very large geographic area. The variability between regions, in damage per unit of pollution (e.g., Mg of sediment), probably is less than the variability among site-specific locations within regions. Each region may contain specific locations where the benefits exceed the costs and other locations where the reverse is true. However, the benefit-cost comparison does suggest four directions for future consideration of BMP implementation: 1. The BMP's recommended or required for specific locations should reflect the characteristics of the site. Treating large geographical areas as homogeneous units in the selection of BMP's may lead to unwanted water quality degradation at some sites and over-spending at other sites. Implementation of BMP's should focus on those forest areas with the greatest potential benefits. 2. Selection of BMP's should be based on the downstream impacts of water quality degradation as well as the more easily observed onsite disturbances. Across sites, the damage per unit of polluting substance that leaves the treatment site varies widely. 3. At specific sites, initial expenditures on BMP's may be most effective. At some point, the marginal benefit of increasingly more stringent controls falls below the marginal cost of their implementation. In any case, more careful economic comparisons should certainly be performed to better understand the marginal costs and benefits of specific BMP's for various types of land areas and management practices. 4. Better estimates of the costs and benefits of BMP use are needed.

[10] Forest activities that increase sediment movement to streams, such as intensive site preparation after harvesting, can alter forest productivity. Early studies that examined very severe treatments, such as windrowing that removed several inches of topsoil, found significant declines in site productivity that produced substantial costs of reduced future timber yields (Dissmeyer et al. 1987). However, more moderate application of the same types of site preparation (such as windrowing only of slash, with minimal topsoil removal) do not appear to decrease productivity (Allen et al. 1991; L. Morris, University of Georgia, personal communication). In all cases, the reduced productivity came from the movement of soil and nutrients into windrows (typically 150 to 200 Mg/ha of soil moved), rather than from erosion losses from the site (typically less than 5 Mg/ha of soil). Therefore, we expect that erosion does not result in any direct cost of reduced productivity onsite, although high rates of erosion may coincide with poor treatments that do affect site productivity (Dissmeyer 1985).

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Appendix

Table A.1 lists selected characteristics for 43 relatively undisturbed forest and rangeland USGS benchmark stations. The map number listed in the table indicates the location of the station as seen in figure A.1.

Table A.2 lists mean annual values for selected water quality parameters for the benchmark stations listed in table A.1.

Tables A.3 - A.8 summarize the findings for key water quality parameters at experimental watersheds in the 6 regions corresponding to Chapters 3-8.

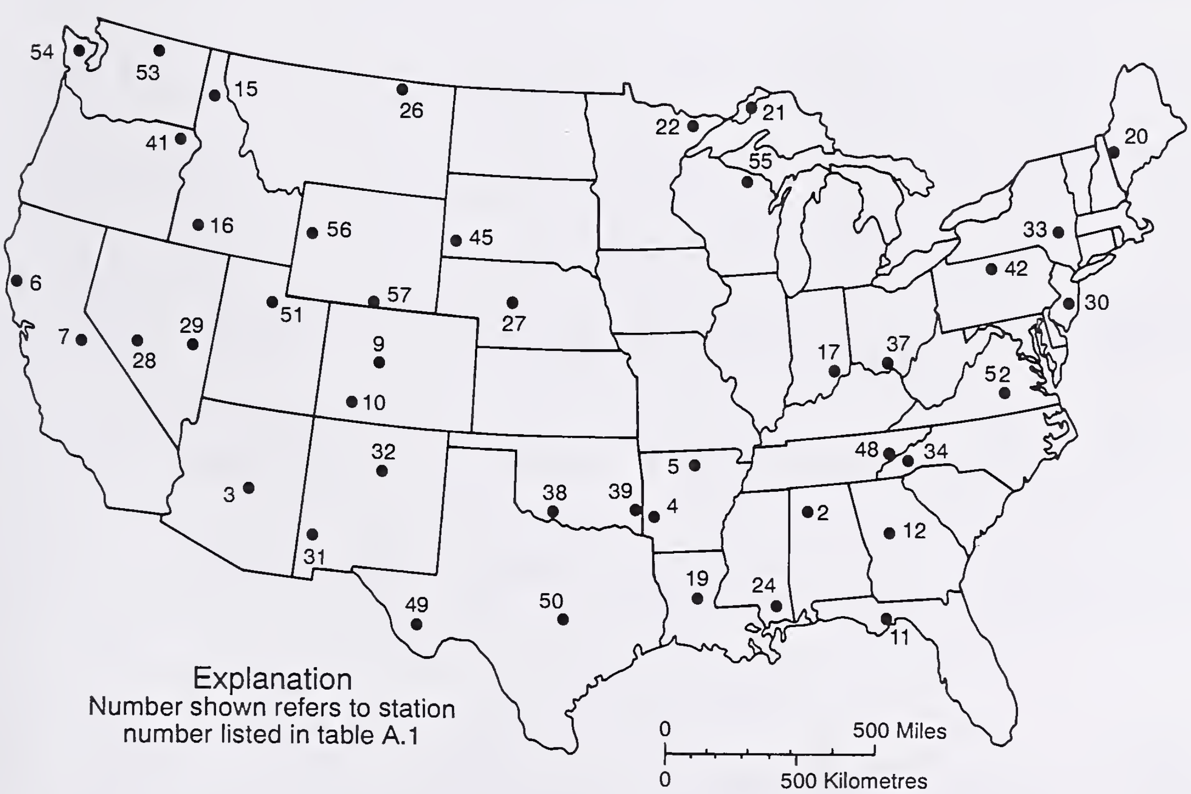

Figure A.1. Locations of hydrologic benchmark stations draining areas largely covered with forest or rangeland vegetation.

Table A.1.—Characteristics of selected hydrologic benchmark stations.[a]

Number	Station name[b]	State	Map number[c]	Drainage (km²)	Flow type[d]	Flow (cm/yr)	Discharge	Precip (m³/sec)	Vegetation (cm/yr)[e]	Influences[f]	Ownership[g]
Southeast											
	Sipsey Fork nr Grayson	AL	2	239	P	66	4.39	132	98% pine and hw	S	NF
	N. Sylamore C nr Fifty Six	AR	5	150	P	48	1.19	114	hw, ug	S	NF
	Cossatot R nr Vandervoort[h]	AR	4	232	P	58	5.58	132	pine & hw, ug	S	NF
	Sopchopp R nr Sopchoppy	FL	11	264	P	64	4.42	142	swamps, pine, oak, ug	S	NF
	Falling C nr Juliette	GA	12	187	P	36	1.50	112	sg pine, hw	S,H	NF
	Big Creek at Pollock	LA	19	132	P	41	1.47	142	sg pine, open land	S,H	NF
	Cypress C nr Janice	MS	24	136	P	51	2.69	152	sg pine, some ag	S,H	M,NF
	Cataloochee C nr Cata.	NC	34	127	P	76	3.14	124	100% hw; ug	R	NP
	Little R above Townsend[h]	TN	48	275	P	91	8.33	147	100% hw, sp, fir	H	NP
	Holiday C nr Andersonville	VA	52	22	P	36	0.20	109	sg hw, sg pine	S	SF
Northeast											
	Wild R at Gilead	ME	20	180	P	86	4.53	112	hw,little conif	S	NF
	McDonalds B Lebanon St Forest	NJ	30	6	P	33	0.03	112	oak, pine,cedar	S	SF
	Esopus C at Shandaken	NY	33	154	P	64	3.79	107	hw, conif	H,R	P
	Young Womans C nr Renovo	PA	42	120	P	43	2.01	97	hw	H	95% public
Midwest											
	S. Hogan C nr Dillsboro	IN	17	99	S	30	1.08	102	pst, some trees nr streams	H	P
	Washington C at Windigo I. Royale	MI	21	34	P	33	0.45	71	hw, sp, fir	R	NP
	Kawishiw R nr Ely	MN	22	655	P	25	6.00	71	pine, lakes	S	NF
	Upper Twin C at McGaw	OH	37	32	P	38	0.34	109	sg hw, some ag	S,F	SF
	Popple R nr Fence	WI	55	360	P	30	3.40	74	95% asp, hw, pine;abd. ag	H,F	P
Great Plains											
	Dismal R nr Thedford	NE	27	2486	P	8	5.38	51	rangeland, few trees	G,F	P
	Blue Beaver C nr Cache	OK	38	64	I	10	0.25	74	native grass, bljack, oak	R	WL,M
	Kiamichi nr Big Cedar	OK	39	104	I	58	2.07	142	pine, hw, grassland	S,H,F	NF
	Castle C ab Dr Res nr Hill Ct	SD	45	215	P	4	0.28	51	90% pine; sp, asp, willow	G,S,F	NF
	S. Fork Rocky C nr Briggs	TX	50	86	I	8	0.37	76	80% gr; 2% ag; oak, willow	H,F,G	P
	Limpia C ab Fort Davis[i]	TX	49	136	I	0.38	0.08	58	cactus, grass, oak, willow	few	?
Rocky Mountains											
	Halfmoon C nr Malta	CO	9	61	P	46	0.74	76	pine, sp, fir, alp.	R	NF
	Vallecito C nr Bayfield	CO	10	187	P	51	3.85	102	sp, aspen, alp	None	W
	Hayden C bl N Fk, nr Hayden	ID	15	57	P	53	0.68	102	sg pine and fir	S	?
	Big Jacks C nr Bruneau (aka Wick)	ID	16	655	I	0.25	0.25	25	Sagebrush and grass	G,B	?
	Rock C blw Horse C nr Int Bndry[h,j]	MT	26	850	I	na	na	na	prairie grasses, sage	few	NG
	S Twin R nr Round Mtn	NV	28	52	P	8	0.14	20	Pinyon, grass, willow	G,M	?
	Steptoe C nr Ely	NV	29	29	P	13	0.17	30	Pinyon and grass	G,R	?
	Red Butte C at Ft. Douglas	UT	51	19	P	25	0.08	64	oak brush, evergreen, asp	few	MW
	Cache C nr Jackson	WY	56	27	P	48	0.34	76	pine, fir, sp, grass, brush	None	?
	Encampment R ab hog pk c nr Enc	WY	57	188	P	43	3.03	76	pine, fir, sp, mead, alp.	R,S	?
Northwest											
	Minam R at Minam	OR	41	622	P	64	12.23	102	pine, fir, larch	None	W
	Andrews C nr Mazama[i]	WA	53	57	P	52	0.94	89	fir, cedar, hemlock, ug	None	PA
	N Fork Quinault R nr Amanda Pk[i]	WA	54	192	P	368	24.50	500	hemlock, fir, sp, cedar, alp	None	NP
Southwest											
	Wet Bottom C nr Childs	AZ	3	94	I	5	0.34	64	chap, pine at high elev	G	W
	Merced R at Happy Isles Br	CA	7	469	P	64	14.75	140	45% bush, fir, pine; alpine	None	NP
	Elder C nr Branscomb	CA	6	17	P	127	0.65	203	dense fir, pines	None	NC
	Mogollon C nr Cliff	NM	31	179	P	9	0.76	33	pine, sp, juniper, willow	None	W
	Rio Moro nr Terrero	NM	32	138	P	18	0.76	61	80% pine, sp, fir, asp; oak	G	W

[a] Sources: (a) Station, name, and drainage area from Hydrologic Benchmark Station List, furnished by Ken Wahl, USGS, Denver 4/5/90; (b) flow type, flow in inches, precipitation, vegetation, and water quality data from Cobb and Biesechker (1971); (c) discharge and precipitation from table 1 of Smith and Alexander (1983). Note: Smith and Alexander took precipitation from Cobb and Biesechker.
[b] U.S. Geological Survey station name.
[c] Except for number 26, these map numbers were used by Cobb and Biesechker.
[d] P=perennial; I=intermittent; S=seasonal.
[e] hw=hardwood; alp=above tree line, sp=spruce, ug=dense undergrowth; sg=second growth.
[f] Influences: B=brush removal; S=silviculture (harvest); H=homes; F=farming; G=grazing; R=recreation; M=mining.
[g] Ownership: W=wilderness; NF=national forest; NP=national park; NC=Nature Conservancy; M=military; IR=Indian reservation; PA=primitive area; MW=municipal watershed; WL=wildlife refuge; SF=state forest; NG=national or provincial grassland; P=private.
[h] Station not used by Smith and Alexander (1983).
[i] Station not on Wahl's list.
[j] Station not included in Cobb and Biesechker (1971).

Table A.2.—Mean annual water quality at selected USGS benchmark stations.

Region	State	Map number	Period of record[a]	Temp (°C)	pH	Conduc (mic/cent)	Bicarbon (mg/l)	Diss sol (mg/l)	Diss O (mg/l)	Diss N (mg/l)	SS (mg/l)
Southeast											
	AL	2	65-86	14.33	7.20	84.98	43.23	49.87	10.25	0.21	22.41
	AR	5	66-88	15.03	8.05	265.22	163.62	149.12	9.85	0.12	7.67
	AR	4	67-88	16.70	7.09	73.12	20.00	27.67	9.09	0.20	3.63
	FL	11	64-88	18.50	5.34	73.43	27.90	42.32	7.90	0.23	6.27
	GA	12	64-87	15.55	7.15	117.49	61.78	79.18	9.26	0.24	22.15
	LA	19	64-88	17.96	6.44	39.70	13.56	39.66	8.8	0.44	36.01
	MS	24	66-86	18.14	6.05	24.44	4.02	23.27	8.90	0.42	36.30
	NC	34	67-86	9.46	6.73	15.12	6.31	16.33	10.54	0.33	8.75
	TN	8	64-87	12.23	6.80	18.30	7.71	14.79	11.00	0.42	15.14
	VA	52	67-88	12.75	6.82	37.25	16.06	32.55	10.62	0.46	6.54
Northeast											
	ME	20	64-86	7.68	6.47	24.17	6.65	19.86	11.56	0.14	4.53
	NJ	30	64-88	10.23	4.18	50.20	0.15	18.24	4.56	0.26	3.94
	NY	33	64-86	8.98	6.87	54.34	13.40	30.37	11.68	0.73	7.50
	PA	42	65-87	8.72	6.82	40.54	9.59	26.24	11.49	0.65	5.66
Midwest											
	IN	17	68-85	12.79	8.01	472.98	205.97	282.03	10.88	2.33	60.39
	MI	21	65-88	7.58	7.42	133.20	75.12	86.66	11.07	0.67	6.37
	MN	2	66-87	10.31	7.02	32.44	13.18	23.21	10.03	0.29	2.85
	OH	37	64-88	13.00	6.89	101.02	15.33	64.60	10.43	1.02	29.56
	WI	55	64-87	8.45	7.36	168.38	95.51	98.78	10.34	1.02	5.75
Great Plains											
	NE	27	66-87	11.42	7.70	175.25	101.45	153.50	9.55	1.28	590.60
	OK	38	65-88	14.60	7.28	165.61	67.46	104.05	9.20	0.38	15.31
	OK	39	65-88	15.94	6.87	26.76	8.51	22.80	9.45	0.27	12.29
	SD	45	64-88	6.10	8.26	467.32	296.35	251.78	10.19	0.51	59.65
	TX	50	64-88	18.74	7.82	469.14	267.32	259.91	8.81	2.31	55.72
	TX	49	67-86	19.36	7.44	164.84	69.11	116.00	9.30	1.55	561.00
Rocky Mountains											
	CO	9	64-85	4.37	7.37	96.27	44.03	48.65	8.98	0.57	7.56
	CO	10	63-86	4.08	7.45	73.93	32.37	44.35	9.79	0.47	4.39
	ID	15	85-88	6.70	7.48	64.90	20.00	50.00	11.35		2.31
	ID	16	85-86	6.69	8.18	153.13	55.00	120.00	10.93		226.00
	MT	26	77-87	8.09	8.28	1150.81	315.55	794.70	9.39	2.25	106.07
	NV	29	68-88	6.99	8.31	317.89	197.52	180.19	9.39	0.64	41.78
	NV	28	67-88	6.76	7.93	122.42	66.10	85.22	9.62	0.30	39.64
	UT	51	64-88	7.32	8.15	593.02	278.30	370.98	9.92	.34	199.91
	WY	56	65-86	3.76	8.28	315.50	208.26	178.82	10.55	0.13	28.23
	WY	57	64-88	5.70	7.33	63.43	32.84	44.47	9.36	0.12	6.56
Pacific Northwest											
	WA	54	65-86	6.10	7.27	73.87	32.25	46.18	12.10	0.09	10.24
	WA	53	71-88	2.99	7.58	47.11	28.98	37.44	11.46		1.66
	OR	41	66-87	6.77	7.40	49.11	30.45	46.54	11.60	0.13	11.53
Pacific Southwest											
	AZ	3	68-88	16.34	7.84	260.29	141.76	170.41	8.68	0.20	5.39
	CA	7	68-88	7.00	6.69	22.28	8.55	20.10	10.92	0.13	2.21
	CA	6	68-88	10.79	7.65	111.03	61.86	73.64	10.42	0.46	11.65
	NM	32	64-88	5.57	7.66	101.25	51.93	61.34	9.61	0.10	28.45
	NM	31	67-88	11.23	7.61	109.42	43.94	80.31	9.36	0.16	14.92
mean				10.51	7.27	163.28	75.79	104.33	9.96	0.56	54.29
st dev				4.62	0.79	205.96	87.17	133.58	1.28	0.59	123.95
min				2.99	4.18	15.12	0.15	14.79	4.56	0.09	1.66
max				19.36	8.31	1150.81	315.55	794.70	12.10	2.33	590.60
coef of var				0.44	0.11	1.26	1.15	1.28	0.13	1.04	2.28

[a] *Beginning and ending year of collection of data for this table. The number of months in any one year during which a sample was taken varies across stations and constituents within the range from 0 to 12.*

Table A.3.—Key water quality parameters for small watersheds in the southeastern United States. Level of detail is quite variable, hence the inconsistencies in entries.

Location	Treatment; watershed area (ha); PPT (mm); runoff (mm)	Period	Suspended sediment (mg/L); Turbidity (NTU)	Temperature pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Coweeta, NC;	Control, WS-2; 12 ha; PPT 1772; Runoff 854	10 yr		pH 3.7	0.003	Ca 0.58 Mg 0.33 Na 1.22 K 0.50 NH_4N 0.002 PO_4P 0.002 SO_4S 0.15 Cl 0.66 TD Si 8.80	Swank and Waide 1988 Swank 1988
	Control WS-27; 39 ha; PPT 2451; Runoff 1737	10 yr		pH 3.5	0.018	Ca 0.36 Mg 0.20 Na 0.48 K 0.23 NH_4N 0.004 PO_4P 0.001 SO_4S 0.38 Cl 0.49	
	White pine, WS-1; 16 ha	10 yr			0.02	Ca 0.651 Mg 0.36 Na 1.06 K 0.52 NH_4N 0.003 PO_4P 0.008 SO_4S 0.14 Cl 0.68 SiO_2 5.42	
	Grass-to-forest succession, WS-6; 9 ha	10 yr			0.67	Ca 0.99 Mg 0.60 Na 1.00 K 0.55 $NH_{4+}N$ 0.005 PO_4P 0.007 $SO_{42-}S$ 0.15 Cl 1.04 SiO_2 6.59	
	Coppice regrowth, WS-7; 59 ha	Calibration			0.00	Ca 0.7-1.35 K 0.0-0.9	
		After treatment			0.01-0.16	Ca 0.75-1.60 K 0.0-0.6	

Table A.3.—Continued

Location	Treatment; watershed area (ha); PPT (mm); runoff (mm)	Period	Suspended sediment (mg/L); Turbidity (NTU)	Temperature pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
	White pine, WS-17; 13 ha	10 yr			0.13	Ca 0.51 Mg 0.23 Na 0.79 K 0.39 NH_4N 0.04 PO_4P 0.002 SO_4S 0.16 Cl 0.53 SiO_2 6.72	
	Coppice regrowth, WS-37; 44 ha				0.18	Ca 0.73 Mg 0.31 Na 0.64 K 0.38 NH_4N 0.004 PO_4P 0.001 SO_4S 0.38 Cl 0.47 SiO_2 4.64	
	Uneven-aged hardwood, WS-40; 20 ha	1 year			0.005	Ca 1.04 Mg 0.40 Na 1.12 K 0.54 NH_4N 0.004 PO_4P 0.003 SO_4S 0.15 Cl 0.61	
	Clearcut + herbicide	First yr		3 °C increases in maximum			Swift and Messer 1971
Great Smoky Mountain National Park	Watersheds at 1500 m; PPT 2500	1 yr	0.3 NTU		4.9	Ca 0.25-1.5 Mg 0.1-0.5 Na 0.75-1.25 K 0.5-1.5 SiO_2 4.3-10.7	Silsbee and Larson 1982
	Watersheds at 500 m; PPT 1400	1 yr	2.5 NTU		0.5		
	< 25% logged	1 yr	0.47 NTU		0.83		
	> 75% logged	1 yr	0.58 NTU		0.36		

Table A.3.—Continued

Location	Treatment; watershed area (ha); PPT (mm); runoff (mm)	Period	Suspended sediment (mg/L); Turbidity (NTU)	Temperature pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Carteret County NC	Forest (700 h))	1985-1988	12.9 mg/L 20 NTU			Total N 1.1 NH_4N+NO_3N 0.1 Total P 0.07 Biological Oxygen Demand 1.3	Hughes et al. 1989
	Forest (24 ha)	March 1989	<1-10 mg/L 1-19 NTU		0-1.0	NH_4N 0-0.2 PO_4P 0.01-0.06 BOD 0.2-0.9 TC˙ <2-8	
Walker Branch, TN	Control; 97.5 ha; PPT 1368; Runoff 713	3 yr			0.057	Ca 16 Mg 8.4 Na 0.48 K 0.73 NH_4N 0.022 PO_4P 0.001 TN 0.156	Elwood and Turner 1989
Santee Experimental Forest, SC	Control and prescribe burned	3 yr			0.02	NH_4N 0.03 PO_4P 0.03	Richter et al. 1982
Georgetown County, SC	Control, hardwood forest	2 yr		pH 4.4	0.47	Ca 5.5 Mg 2.7 K 0.80 NH_4N 0.047 SO_4S 10.95 TP <0.01 DO 5.1	Askew and Williams 1986
	Drained	2 yr		pH 4.9	0.94	Ca 10.8 Mg 3.5 K 0.87 NH_4N 0.074 SO_4S 15.6 TP <0.01 DO 5.0	
	Logged	2 yr		pH 5.2	0.11	Ca 7.9 Mg 2.6 K 2.15 NH_4N 0.057 SO_4S 10.08 TP <0.01 DO 5.8	

Table A.3.—Continued

Location	Treatment; watershed area (ha); PPT (mm); runoff (mm)	Period	Suspended sediment (mg/L); Turbidity (NTU)	Temperature pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
	Site prepared	2 yr		pH 5.5	0.20	Ca 6.0 Mg 2.2 K 1.6 NH_4N 0.026 SO_4-S 7.91 TP <0.01 DO 5.7	
	Young pine forest	2 yr		pH 5.8	0.05	Ca 5.3 Mg 2.3 K 2.0 NH_4N 0.022 SO_4S 6.21 TP <0.01 DO 5.4	
	Old pine forest	2 yr		pH 5.4	0.48	Ca 5.5 Mg 2.2 K 1.5 NH_4N 0.041 SO_4S 7.48 TP <0.01 DO 6.9	
Bradford County, FL	Control, pine forest; 137 ha; PPT 1400; Runoff 583	3 yr	3 mg/L (annual average)	pH 3.8	0.03	Ca 0.60 Mg 0.73 K 0.14 NH_4N 0.13 PO_4P 0.02 TKN 1.15 TP 0.02	Riekerk 1983
	Harvest, minimum impact; 64 ha; PPT 1400; Runoff 618	2 post-treat yr	4 mg/L	pH 3.6	0.04	Ca 0.20-0.38 Mg 0.77-1.43 K 0.25-0.78 NH_4N 0.06-0.15 PO_4P 0.00-0.03 TKN 1.13-1.45 TP 0.02-0.05	

Table A.3.—Continued

Location	Treatment; watershed area (ha); PPT (mm); runoff (mm)	Period	Suspended sediment (mg/L); Turbidity (NTU)	Temperature pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
	Harvest, maximum impact; 48 ha; PPT 1400; Runoff 656	2 post-treat yr	13 mg/L	pH 4.1	0.05	Ca 0.89-1.69 Mg 0.66-1.71 K 0.45-0.71 NH_4N 0.08-0.09 PO_4P 0.00-0.01 TKN 0.87-1.05 TP 0.01-0.02	
Grant Forest, GA	Control, pine forest, 42.5 ha	6 yr			0.116-0.156	Ca 5.96-6.55 Mg 2.56-2.84 Na 5.55-6.46 K 1.28-2.53 TP 0.23-0.61	Hewlett et al. 1984 Hewlett et al. 1984 Hewlett and Fortson 1982
	Clearcut, 32.5 ha	Calibration, 1 yr			0.043	Ca 3.49 Mg 1.33 Na 5.59 K 2.50 TP 0.23	
		2 yr		11 °C maximum increase in summer; 6 °C maximum decrease in winter	0.027	Ca 2.89 Mg 1.38 Na 4.49 K 1.32 TP 0.32	
	Recovery period	3 yr			0.045	Ca 3.38 Mg 1.43 Na 4.49 K 1.34 TP 0.55	
Natchez Trace State Forest, TN	Controls, pine forests	3 yr	Stormflow 82				McClurkin et al. 1985
	Harvested	1-3 yr post harvest	Stormflow 183				
N Mississippi P. taeda P. elliotti (5 watersheds)	1.49-2.81 ha 135 cm rainfall		Stormflow susp.sed. 49-228		Stormflow 0.01-0.02	Stormflow NH_4N 0.17-0.23 TP 0.024-0.028 COD 20-45 TOC 6-16	Schreiber and Duffy 1982

Table A.3.—Continued

Location	Treatment; watershed area (ha); PPT (mm); runoff (mm)	Period	Suspended sediment (mg/L); Turbidity (NTU)	Temperature pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Upper coastal plain, MS	Control, hardwood/pine forest	1976 1977	2127 mg/L 393 mg/L				Beasley 1979
	Harvest, chopped	1976 1977	2471 mg/L 670 mg/L				
	Harvest, sheared	1976 1977	2837 mg/L 794 mg/L				
	Harvest, bedded	1976 1977	2808 mg/L 2346 mg/L				

* TC: Total coliforms (*/100 ml)

Table A.4.—Key water quality parameters for small watersheds in the northeastern U.S. and eastern Canada.

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Hubbard Brook, NH	Control (W4-W6) hardwood forest	1965-1968	25 kg/ha/yr stream	0.09 NTU		0.2	Ca 1.27-1.80 Mg 0.35-0.41 Na 0.80-1.13 K 0.18-0.26 NH_4N 0.02-0.09 SO_4S 2.00-2.14 Cl 0.55-0.58 SiO_2 3.8-5.5 Al 0.12-0.33 [W3] DO(%) 30-100	Likens et al. 1970 Hornbeck et al. 1987 Lawrence and Driscoll 1988
	Devegetated (W2)	1965-1966			4 °C increased maximum	0.21	Ca 1.81 Mg 0.37 Na 0.87 K 0.19 NH_4N 0.14 SO_4S 2.27 Cl 0.54 SiO_2 4.1 Al 0.22	
		1966-1967	x4 pre-treatment	0-0.3 NTU	1 °C increased maximum	8.7	Ca 6.45 Mg 1.35 Na 1.51 K 1.92 NH_4N 0.05 SO_4S 1.27 Cl 0.89 SiO_2 5.6 Al 1.5	
		1967-1968	x4 pre-treatment			11.9	Ca 7.55 Mg 1.55 Na 1.54 K 2.96 NH_4N 0.04 SO_4S 1.23 Cl 0.75 SiO_2 5.77 Al 2.0	
	Stripcut	yr 1				1.4 (fig. 4.2)	Ca 2.00 Mg 0.43 Na 1.22 K 0.23 NH_4N 0.028 SO_4S 2.13 Cl 0.57	

Table A.4.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
	Blockcut	yr 1				3.9 (fig. 4.2)	Ca 1.28 Mg 0.41 Na 0.9 K 0.20 NH_4N 0.014 SO_4S 1.84 Cl 0.50	
	Whole-tree harvest	yr 1				3.6 (fig. 4.2)	SO_4S 1.76 Al 0.81	
Nashwaak River, NB	Control, hardwood/ conifer forest	4 yr				0.12		Krause 1982
	Harvested	1-3 yr post harvest				0.6 average; 1.3 maximum		
Various locations, NE	Control and harvested, central hardwoods	2 yrs				0	Ca 1-5 Mg 0.3-1 Na 1.5-2.5 K 0.2-1.2 SO_4S 3-5.2 Cl 1.5-2.5	Martin et al. 1984
	Control and harvested, conifer forests	2 yr				0.1 to 0.5	Ca 1.5-21 Mg 0.5-3 Na 0.5-2.5 K 0-1 SO_4S 1-3 Cl 0.5-2.5	
	Control and harvested, northern hardwoods	2 yr				0.1 to 2.0	Ca 1-7 Mg 0.2-1.5 Na 0.3-1.5 K 0.1-1.0 SO_4S 1-3 Cl 0.1-1.0	
Leading Ridge, PA	Control, mixed hardwood forest	3 yr		1.7 mg/L 2 NTU		0.03	Ca 5.81 Mg 2.28 Na 1.03 K 0.96 SO_4S 2.67	Lynch et al. 1975 Lynch and Corbett 1990 Lynch et al. 1985
	43% of watershed harvested	1-3 yr past harvest		5.9 mg/L 3 NTU		0.08 average; 0.85 maximum	Ca 3.15 Mg 1.49 Na 0.90 K 1.13 SO_4S 1.9	

Table A.4.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
	Devegetated	2 yrs past havest		80 mg/L		2.5	Ca 3.15 Mg 1.76 Na 0.93 K 1.57 SO_4S 1.29	
Quaker Run, PA	Upstream	1985-1986			pH 5	0.13	Ca 2 Mg 2.7 Na 1.1 K 0.6 SiO_2 6.3	Phillips and Stewart 1990
	Downstream	1985-1986			pH 7	0.32	Ca 6.5 Mg 2.0 Na 2.5 K 0.9 SiO_2 6.5	
Fernow Experimental Forest, WV	Control, mixed hardwood forest	3 yr	17 kg/ha/ to stream	2.1 NTU Dis. solids 12.2	14.4 °C pH 6.0	0.2	Ca 0.5-1.5 Mg 0.2-0.5 Na 0.4-0.8 K 0.3-0.7 NH_4N 0-0.5 PO_4P 0.026-0.065 SO_4S 0.67-1.67	Aubertin and Patric 1974
	Harvested	3 yr	49 kg ha^{-1} to stream	3.1 NTU Dis. Solids 11.5	15.6 °C pH 6.0	0.2 average; 1.4 maximum	Ca 0.5-1.5 Mg 0.2-0.5 Na 0.4-1.4 K 0.3-0.7 NH_4N 0-0.5 PO_4P 0-0.029 SO_4S 0.33-1.67	
	Fertilized	3 yr				> 10 for 3 weeks		Edwards et al. 1991

Table A.5.—Key water quality parameters for the North Central and Great Plains regions.

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Marcell Experimental Forest, MN	Control, aspen forest	2 yr				0.3	Ca 3.0 Mg 1.3 Na 0.8 K 1.5 NH_4N 0.41 PO_4P 0.039 Cl 0.8 TKN 0.85	Verry 1972
	Harvested	2 yr				0.015	Ca 2.7 Mg 1.0 Na 0.7 K 1.5 NH_4N 0.55 PO_4P 0.055 Cl 0.5 TKN 0.8	
Cherokee County, TX	Control, pine forest	Calibration (1980)	340	71 NTU	pH 5.4	0.007	Ca 1.8 Mg 0.7 Na 0.7 K 1.7 NH_4N 0.51 PO_4P 0.0003 TP 0.099	Blackburn et al. 1986 Blackburn and Woods 1990
		First year	33	140 NTU 112 mg/L	pH 5.9	0.002	Ca 1.7 Mg 1.1 Na 1.0 K 2.8 NH_4N 0.036 PO_4P 0.0033 TP 0.077	
		Second year	5	61 NTU 79 mg/L	pH 5.4	0.002	Ca 1.0 Mg 0.8 Na 1.1 K 2.0 NH_4N 0.040 PO_4P 0.0039 TP 0.039	
		Third year	5	38 NTU 31 mg/L	pH 6.6	0.002	Ca 0.7 Mg 1.0 Na 1.2 K 2.0 NH_4N 0.015 PO_4P 0.0013 TP 0.030	

Table A.5.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
		Fourth year	29	54 NTU 213 mg/L	pH 6.4	0.015	Ca 0.7 Mg 1.0 Na 0.8 K 2.1 NH_4N 0.071 PO_4P 0.0042 TP 0.092	
		Fifth year		39 NTU	pH 6.2	0.035	Ca 3.0 Mg 1.1 Na 3.8 K 1.9 NH_4N 0.059 PO_4P 0.0026 TP 0.053	
	Harvested, chopped, burned	Calibration (1980)	85	31 NTU	pH 5.5	0.003	Ca 2.3 Mg 0.9 Na 1.0 K 2.1 NH_4N 0.060 PO_4P 0.0003 TP 0.060	
		First year	25	30 mg/L 59 NTU	pH 6.1	0.021	Ca 2.1 Mg 1.3 Na 1.7 K 5.6 NH_4N 0.043 PO_4P 0.0049 TP 0.053	
		Second year	6	36 mg/L 16 NTU	pH 5.6	0.003	Ca 1.1 Mg 0.8 Na 1.4 K 1.9 NH_4N 0.009 PO_4P 0.0013 TP 0.024	
		Third year	5	12 mg/L 16 NTU	pH 6.5	0.004	Ca 0.8 Mg 1.1 Na 1.7 K 3.0 NH_4N 0.016 PO_4P 0.0007 TP 0.027	

Table A.5.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
		Fourth year	16	28 mg/L 17 NTU	pH 6.5	0.028	Ca 1.7 Mg 1.2 Na 3.0 K 3.3 NH_4N 0.068 PO_4P 0.0042 TP 0.043	
		Fifth year		21 NTU	pH 6.2	0.023	Ca 1.4 Mg 0.9 Na 1.7 K 4.2 NH_4N 0.035 PO_4P 0.0016 TP 0.038	
	Harvested, sheared, windrowed, burned	Calibration	184	79 NTU	pH 5.3	0.003	Ca 2.4 Mg 0.7 Na 0.7 K 3.2 NH_4N 0.067 PO_4P 0.0003 TP 0.068	
		First year	2940	1158 mg/L 153 NTU	pH 6.0	0.046	Ca 0.9 Mg 1.4 Na 1.4 K 4.9 NH_4N 0.058 PO_4P 0.0088 TP 0.219	
		Second year	80	256 mg/L 60 NTU	pH 5.8	0.014	Ca 2.9 Mg 1.1 Na 2.2 K 3.2 NH_4N 0.022 PO_4P 0.0020 TP 0.055	
		Third year	35	113 mg/L 47 NTU	pH 6.5	0.011	Ca 0.8 Mg 0.9 Na 1.4 K 3.0 NH_4N 0.022 PO_4P 0.0003 TP 0.033	

Table A.5.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
		Fourth year	165	489 mg/L 61 NTU	pH 6.3	0.010	Ca 1.4 Mg 1.2 Na 3.2 K 4.5 NH_4N 0.033 PO_4P 0.0020 TP 0.058	
		Fifth year		38 NTU	pH 6.3	0.011	Ca 1.4 Mg 0.9 Na 3.2 K 3.7 NH_4N 0.090 PO_4P 0.0033 TP 0.049	

Table A.6.—Key water quality parameters for the Rocky Mountains.

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Fraser Experimental Forest, CO	Fool Creek, road construction, partial cutting	1950-1965		< 5 mg/L				Alexander et al. 1985 Stottlemyer 1987 Leaf 1975
		1952-1966	99 to stream					
		1984				0.002	Ca 5.95 Mg 1.03 Na 1.93 K 1.25 NH_4N 0.014 PO_4P 0.0024 SO_4S 0.61 Cl 0.67	
	Control, Lexen Creek	1982-1986				0.006		
		1955-1966	36 to stream					
		1984					Ca 10.64 Mg 1.19 Na 1.01 K 0.51 NH_4N 0.028 SO_4S 0.63	
	Deadhorse Creek, road construction and harvesting	1982-1986				0.06		
		1956-1966	25 to stream					
		1984					Ca 17.64 Mg 2.09 Na 1.59 K 1.09 NH_4N 0.0042 SO_4S 0.83	
Silver Creek, ID	Control, ponderosa pine/ Douglas-fir	1975-1981				0.01		Clayton and Kennedy 1985
	Harvested	1977-1979				0.018 average 0.05 peak		
		1981				0.01		
Priest River, ID	Benton Creek, above cut	1 yr		4.5 mg/L		0.17		Snyder et al. 1975
	In cut unit			6.4 mg/L		1.5		
	Below cut unit			5.0 mg/L		0.2		

Table A.6.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
	Ida Creek, above cut	1 yr		7.1 mg/L		0.15		
	In cut unit			37 mg/L		0.15		
	Below cut unit			12 mg/L		0.15		
	Canyon Creek, above cut	1 yr		3 mg/L		0.015		
	In cut unit			16 mg/L		0.18		
	Below cut unit			4 mg/L		0.015		
Bitterroot National Forest, MT	Spruce Creek, control	Third year post harvest				0.11		Bateridge 1974
	Lodgepole Creek, harvested					0.19		
	Springer Creek, control	First year post harvest				0.17		
	Mink Creek, harvested					0.13		
	Little Mink Creek, control	First year post harvest				0.17		
	Harvested					0.4		
West central Alberta	Control and harvested forests of pine, spruce, aspen	Snowmelt and summer recession flows				0.005 to 0.05		Singh and Kalra 1975
Chicken Creek, UT	Control, aspen					0.008		Johnston 1984
	Harvested					0.025		
Manitou Experimental Forest, CO	Ungrazed and grazed pastures	1 summer		35 to 65 mg/L		0.03-0.07	FC* 200 ungrazed FC 1050 grazed FS** 730 ungrazed FS 1760 grazed	Johnson et al. 1978
S. Fork Poudre River, CO	Ungrazed Little Beaver Creek	2 yr		7 to 25 mg/L			TC*** 37 FC 4 FS 14	Meiman and Kunkle 1967
	Grazed Pennock Creek	2 yr		4 to 21 mg/L			TC 120 FC 68 FS 24	
Nash Fork, WY	Low-use natural area	1 yr					FC 0.2 to 1.2	Skinner et al.
	Grazed, recreational watersheds	1 yr					FC 20 to 30	1974

Table A.6.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Reynolds Creek, ID	Grazed sagebrush watersheds						FC maximum 2500	Stephenson and Street 1978

*Fecal coliform; **Fecal strepotocci; ***Total coliform colonies/100 mL.

Table A.7.—Key water quality parameters for the Pacific Northwest.

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
HJ Andrews Experimental Forest, OR	Control, WS-2			3 mg/L annual average Winter 1966/67 > 10 mg/L 9 days		0.01	PO_4P 0.005-0.01	Fredriksen et al. 1975
	Harvested and burned, WS-1			Winter 1966/67 > 10 mg/L 26 days >1000 mg/L 2 days	Summer maximum increased by 8 °C	0.4 mg/L peak	PO_4P 0.008-0.013	
	Road construction, 25% harvested, WS-3	First 2 road years Third year Third year post harvest		15 mg/L 260 mg/L 2500 mg/L				
	Control, WS-9	Calibration year First year Second year Third year Fourth year	(*) Input to channel: SUR 26 DIS 16 Export from channel: DIS 286 SUS 35 BED 14			0.001 0.001 0.004 0.002 0.008	TKN 0.061 TKN 0.064 TKN 0.072 DOC 1.3 TKN 0.072 DOC 2.8 NH_4N 0.014 TKN 0.069 DOC 1.3 NH_4N 0.014	Sollins and McCorrison 1981 Sollins et al. 1980
	Clearcut, WS-10	Calibration year	Input to channel: SUR 80 DBS 643 DIS 16 Export from channel: DIS 332 SUS 70 BED 90 DBF 493			0.001	Ca 3.20 Mg 0.834 Na 1.96 K 0.339 TKN 0.047 TP 0.054	
		First year Second year	Input to channel: SUR 199 DBS 1300 DIS 17			0.003 0.067	TKN 0.057 TKN 0.082 DOC 2.7	

Table A.7.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
		Third year	Export from channel: DIS 354 SUS 320 BED 305 DBF 6000			0.033	TKN 0.079 DOC 3.2 NH_4N 0.027	
		Fourth year				0.017	TKN 0.071 DOC 1.3 NH_4N 0.017	
	(W8) Control 21.4 ha	Pre-treatment (2-3 y)		7.94	pH 7.3	0.001	Ca 2.40 Mg 0.64 Na 2.30 K 0.36 TKN 0.046 PO_4P 0.021 $SiO_{2\ 9.03}$	Martin and Harr 1989
		Post-treatment (9 y)		8.2	pH 7.3	0.003	Ca 3.04 Mg 0.57 Na 2.51 K 0.43 TKN 0.039 PO_4P 0.022 SiO_2 12.78	
	(W7) Shelterwood 15.4 ha	Pre-treatment (2-3 y)		2.13	pH 7.3	0.001	Ca 2.70 Mg 0.80 Na 1.76 K 0.41 TKN 0.029 PO_4P 0.022 SiO_2 9.80	
		Post-treatment (9 y)		2.14	pH 7.3	0.006	Ca 3.58 Mg 0.93 Na 2.01 K 0.56 TKN 0.029 PO_4P 0.022 SiO_2 15.15	
	(W6) Clearcut 13.0 ha	Pre-treatment (2-3 y)		4.19	pH 7.3	0.001	Ca 3.05 Mg 0.86 Na 1.80 K 0.29 TKN 0.026 PO_4P 0.016 $SiO_{2\ 10.61}$	

Table A.7.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
		Post-treatment (9 y)		4.24	pH 7.3	0.020	Ca 3.54 Mg 0.89 Na 1.90 K 0.40 TKN 0.034 PO_4P 0.014 SiO_2 13.89	
Middle Fork Santiam River, OR	Above study area	9 year averages		71 mg/L; <30 NTU				Sullivan 1985
	Below study area			50 mg/L; <30 NTU				
Alsea, OR	Control, Flynn Creek	1966-1970		1.5-6.6 mg/L		1.2		Fredriksen et al. 1975, Brown et al. 1973 Brown and
	25% harvested, Deer Creek	1967-1970, post road/harvest		2.0-16.3 mg/L		1.2		
	100% harvested, Needle Branch	1966-1970, post road/harvest		4.4-15.8 mg/L	2 °C increase winter 8 °C increase summer	0.4		Krygier 1970
Coast Range, OR	Control, Siletz Creek	2 yr				0.6	Ca 1.6 Mg 1.0 Na 10.6-11.7 K 0.8	Miller and Newton 1983
	Harvested+Herbicide, Siletz Creek	2 yr				0.6	Ca 1.6 Mg 1.2 Na 11.3-12.2 K 0.8-1.2	
	Control, Drift Creek	2 years				1.5 to 2.0	Ca 2-3.3 Mg 1.9-2.4 Na 12.2-17.3 K 1.2	
	Harvested+Burned+ Herbicide, Drift Creek	2 yr				1.5 to 2.0	Ca 2-3.2 Mg 1.9-2.2 Na 12.4-14.5 K 1.6	
	Control, Brush Creek	2 yr				0.7 to 2.1	Ca 2.8-4.8 Mg 1.7-2.9 Na 11.3-13.3 K 0.8-1.2	
	Harvested+Herbicide, Brush Creek	2 yr				0.7 to 2.1	Ca 2.4-6 Mg 1.7-4.1 Na 8.7-14.7 K 1.6	

Table A.7.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Bull Run, OR	Control, Fox Creek	1970-1981		0.6-2.4 mg/L		0.010	Ca 0.4-1.3 Mg 0.4-0.7 Na 0.6-1.2 K 0-0.3 TN 0.042 PO_4P <0.003 TP 0.014-0.025 SiO_2 1-3	Fredriksen et al. 1975 Fredriksen and Harr 1988 Harr and Fredriksen 1988
	25% harvested, Fox Creek	1973-1981 post harvest		unchanged		0.08 average 0.28 maximum	Ca 0.5-1.3 Mg 0.4-0.8 Na 0.6-1.2 K 0-0.3 TN 0.038 PO_4P <0.004 TP 0.014-0.028 SiO_2 1-3	
	25% harvested and burned, Fox Creek	1971-1981		unchanged		0.04, declining to 0.01 after 5 years	Ca 0.3-1.2 Mg 0.4-0.7 Na 0.7-1.2 K 0-0.3 TN 0.038 PO_4P <0.003 TP 0.014-0.024 SiO_2 1-3	
Coyote Creek, OR	Control			< 40 mg/L		0.015-0.040	NH_4N 0.005	Fredriksen et al.
	Shelterwood			< 40 mg/L	No change	< 0.015		1975
	Patchcuts			< 40 mg/L	No change	0.015-0.040		Harr et al. 1979
	Clearcut	First year Later years		170 mg/L < 40 mg/L	8 °C maximum increase; after 8 years, 3 C maximum increase	0.1		Adams and Stack 1989
UBC Research Forest, BC	Control	1972-1982			<17 °C pH 6.5-6.8	0.015-0.07	Ca 1.14-2.07 Mg 0.2-0.36 Na 0.64-1.06 K 0.06-0.1 NH_4N 0-0.002 SO_4S 0.47-0.99 SiO_2 2.79-5.51 Cl 0.69-1.05	Feller and Kimmins 1984 Feller 1981
	Clearcut	First year			pH 6.4	0.5	Ca 1.58 Mg 0.36	

Table A.7.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
		Third year			pH 6.5 <21.8 °C	Control levels	Na 1.26 K 0.38 NH_4N 0 SO_4S 0.73 SiO_2 6.75 Cl 1.15 Ca 1.64 Mg 0.26 Na 1.00 K 0.22 NH_4N 0.66 SO_4S 0.51 SiO_2 5.65 Cl 0.66	
	Clearcut, burned	First year Third year			pH 6.5 pH 6.5 <20.3 °C	0.17 Control levels	Ca 2.08 Mg 0.25 Na 0.82 K 0.16 NH_4N 0 SO_4S 0.54 SiO_2 2.95 Cl 0.76 Ca 1.14 Mg 0.29 Na 0.96 K 0.09 NH_4N 0 SO_4S 0.51 SiO_2 5.55 Cl 0.77	
Okanagan Valley, BC	Control	1 yr				0.03 average 0.12 maximum		Hetherington 1976
	Harvested	1 yr				0.03 average 0.4 maximum		
High Ridge Watersheds, OR	Control, mixed conifer forest	Pre-treatment Post-treatment			pH 7.1	0.092 0.162	Ca 3.24 Mg 1.20 Na 1.76 K 0.84 TN 0.11 PO_4P 0.025 Ca 3.42 Mg 1.24	Tiedemann et al. 1988

Table A.7.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
							K 1.04 TN 0.13 PO_4P 0.33	
	Selection harvest	Pre-treatment			pH 6.7	0.003	Ca 2.66 Mg 0.82 Na 1.74 K 0.68 TN 0.06 PO_4P 0.014	
		Post treatment (3 y)				0.006	Ca 2.65 Mg 0.88 K 0.88 TN 0.09 PO_4P 0.013	
	Patch cuts	Pre-treatment			pH 7.0	0.001	Ca 2.04 Mg 0.72 Na 1.58 K 0.59 TN 0.02 PO_4P 0.009	
		Post treatment (3 y)				0.004	Ca 2.16 Mg 0.71 K 0.60 TN 0.06 PO_4P 0.008	
	Clearcut	Pre-treatment			pH 7.1	0.004	Ca 2.46 Mg 0.72 Na 1.65 K 0.68 TN 0.003 PO_4P 0.014	
		Post treatment				0.026	Ca 2.74 Mg 0.86 K 0.78 TN 0.08 PO_4P 0.013	
Hansel Creek, WA	Prior to treatment			3.7 mg/L; 1.0 NTU		0.15		Fowler et al. 1988
	After road construction	First year		178 mg/L; 24 NTU		No increase from roads or harvesting		
		Second year		8.5 mg/L; 1.3 NTU				
		Third year		2.1 mg/L; 1.0 NTU				

Table A.7.—Continued

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Maybeso Experimental Forest, AK	Harvested	First few years after harvest			0.5 to 2.0 °C maximum increase			Meehan et al. 1969
Southeastern, AK	Control			0.2-310 mg/L; <1 NTU				Stednick et al. 1982
	Harvested, burned			0.2-1290 mg/L; <5 NTU				
Various locations	Nitrogen fertilization					0.2 to 10		Various studies

(*) SUR: Surface erosion, DIS: Dissolved load, SUS: Suspended load, BED: Bedload, DBS: debris slide, DBF: Debris flow

Table A.8.—Key water quality parameters for the Pacific Southwest.

Location	Treatment; water area (ha); PPT (mm); runoff (mm)	Period	Erosion (kg/ha/yr)	Suspended sediment (mg/L); Turbidity (NTU)	Temperature; pH	Nitrate-N (mg/L)	Other chemicals (mg/L)	Reference
Three Bar Watersheds, Arizona	Control, chaparral, WS-D	15 yr 3 yr			pH 7.9-8.1	< 0.3 mg/L Ca 22-27 Mg 6-7 Na 14-17 K 0.34-0.43 SO_4S 2.67-7.01 Cl 4-6.8		Davis 1984, 1987a,b Davis 1989
	Conversion to grassland, WS-B	After first stage 10 years after second stage				2.7 average 9.5 maximum 3.5 to 11.9 average 18.8 maximum		
	Conversion to grassland, WS-F	Post-treatment (10 yr) Post-treatment (prescribed burn) (1 yr)				< 7.4 average	Ca 20 Mg 6 Na 16 K 0.71 SO_4S 4.0 Cl 5.5	
Beaver Creek Watersheds, AZ	Control, ponderosa pine forest	8 yr				0.01		M. Ryan, personal communication
	Heavy thinning	8 yr				0.05		
	Clearcut	8 yr				0.22		
Castle Creek, AZ	Prescribed fire in ponderosa pine	1 yr				< 0.003 mg/L		Gottfried and DeBano 1990
San Dimas Experimental Forest, CA	Grassland, chaparral	2 yr				2.8 mg/L maximum		Riggan et al. 1985
Southern CA	Chaparral	1 yr				1.3 to 19.6		Riggan et al. 1985

☆U.S. GOVERNMENT PRINTING OFFICE: 1993-574-029/05018

Binkley, Dan; Brown, Thomas C. 1993. Management Impacts on Water Quality of Forests and Rangelands. Gen. Tech. Rep. 239. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 114 p.

Following an overview of water quality concerns in Chapter 1 and a description of basic forest and rangeland hydrology and water quality processes in Chapter 2, we summarize what has been learned about the effects of land management practices on nonpoint source pollution at generally small experimental forest and rangeland sites in the United States and Canada. The final chapter describes laws affecting forest and rangeland water quality and the use of best management practices to protect water quality. The quality of water from forested watersheds is typically very good, even on disturbed and managed sites. At most sites, forest practices lead to minimal impacts on water quality and do not seriously impair fish habitat or water supplies. However, at more sensitive sites, special care is required. Existing best management practices, if followed, adequately protect forest water quality. Most states have active programs to promote the use of such practices. Unlike forest practices, impacts of grazing on water quality have received little careful study, and few states have specified best management practices to control grazing impacts. Future study is needed to improve the specification of best management practices for site specific forest situations. A sufficient base of information is also needed to design efficient best management practices for rangelands.

Keywords: water quality, nonpoint source pollution, forest management, harvest impacts, grazing, best management practices